Hydrography, ecology and water quality management of the South Docks, Liverpool.

Thesis submitted in accordance with the requirements of the University of Liverpool for the degree of Doctor in Philosophy.

by

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ABSTRACT

Changes in shipping practices have led to the decline and disuse of numerous docks in the traditional ports in the estuaries of the U.K. These are now the focus of various redevelopment schemes involving housing, tourism, and recreational use. Water quality is a major impediment to the successful implementation of urban renewal in obsolete docks. In this thesis the South Dock complex in Liverpool is described as a case study of a lower estuarine, redeveloped dock. The source of water for the South Docks is the Mersey Estuary, which suffers from severe pollution problems. The hydrography and ecology of the South Docks were studied and the possible implications for water quality considered. Various methods of improving water quality were investigated. These included the use of an air lift water mixer and a large population of the mussel *Mytilus edulis* L., introduced as a biofilter, in an experimental dock.

Monitoring of physico-chemical and biological parameters was carried out at three sites: the Graving Dock (9-10m deep, used as the experimental dock), Albert Dock (6m deep) and Queens Dock (3-4m deep). The water in the docks was found to be of relatively high salinity (23-28 °/oo), nutrient rich and prone to large annual fluctuations in temperature. Thermal stratification was often recorded in summer in the deeper docks and this was frequently associated with depletion of hypolimnetic dissolved oxygen. Initially, low oxygen concentrations resulted in mortality of benthic fauna and the release of foul smelling gases, but improvements in water quality were seen with time.

Dense phytoplankton communities were able to develop due to the abundant nutrient supply. This detracted from the appearance of the water and caused depletion of dissolved oxygen as the blooms decayed. The presence of 'red tides' of potentially toxic species of dinoflagellates at certain times of the year prompted concern for the health of participants in watersports. The spring to summer succession of phytoplankton was typified by an assemblage of diatoms in early spring, followed by the colonial flagellate *Phyaeocystis pouchettii*. Summer phytoplankton was dominated by dinoflagellates, especially in the shallower docks. Phytoplankton biomass in Queens Dock was higher in summer 1989 and 1990 than in 1988. In the Albert and Graving Docks phytoplankton biomass was greatest in summer 1988. The zooplankton of the South Docks was typically estuarine in character, although lamellibranch larvae were notably sparse. A dramatic decrease in the concentrations of zooplankton in the summers of 1989 and 1990 compared to 1988 was seen in all docks.

At the start of the research project the fauna and flora of the dock walls and sediment was very impoverished. Over the following three years a relatively diverse flora and fauna developed on the walls of the docks. At the start of the research, in June 1988, bryozoa (mainly *Conopeum seurati*) dominated the dock walls and few other species were present. A dense settlement of *Mytilus edulis* occurred on the walls in autumn 1988. Increasing depth penetration of macroalgae was seen with time. At the end of the research project the dock walls were still largely dominated by *Mytilus edulis*, but increasing cover by ascidians and the sponge *Halichondria panicea* was apparent. Evidence of increased colonisation of the sediments was seen towards the end of the study. At the start of the research project the dock ecosystem was dominated by the plankton. As time progressed the benthos became

increasingly important. Filter feeders, particularly *Mytilus edulis* were considered to be an important link between the benthic and pelagic systems.

In the experimental Graving Dock the use of a water mixer eliminated thermal stratification and increased hypolimnetic dissolved oxygen. Mixing alone did not immediately reduce phytoplankton biomass. Reduced oxygen concentrations in deeper layers still occurred with mixing and the mixer was considered to be underpowered for the size of the dock. Biological filtration by mussels (both natural and introduced populations) resulted in dramatic improvements in water clarity and reductions in phytoplankton biomass in the Albert and Graving Docks. Increased dissolved oxygen concentrations and reduced thermal stratification were seen in the Albert Dock which may also have been a result of the reductions in phytoplankton biomass. The potential deleterious effects of dense mussel cultivation are discussed. It is likely that the use of an artificial mixer would eliminate many of the associated problems. The use of a mixer may also improve the efficiency of mussel filtration. The evidence for the existence of two alternative stable states, a turbid and a clear water state, in the South Docks, is discussed, along with possible feedback mechanisms.

The water quality of higher salinity docks can be improved with appropriate management. Such docks may represent a valuable resource for recreation, education and nature conservation in urban areas

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CHAPTER ONE

GENERAL INTRODUCTION

1.1 INTRODUCTION

Marine, brackish and freshwater docks that are no longer used for commercial shipping are common throughout the U.K. (see Fig. 1.1). Such docks are a legacy of developments in world trade and transport systems since the early 18th century. Rapidly increasing maritime trade towards the end of the 17th century had led to the development of more complex quays and warehouses in estuaries and harbours. Cargo handling operations and ship sizes were restricted in many major ports due to large tidal ranges. The turn of the 17th century saw a move towards the construction of enclosed basins with lock gates which retained water and provided a quay-side berth for trading ships throughout the tidal cycle. The building of the world's first purely commercial dock commenced at Liverpool in 1710 (Ritchie-Noakes 1984). Such dock basins, surrounded by warehouses, forming self-contained units, became the standard pattern in the major British ports throughout the 19th and early 20th centuries.

Throughout the Industrial Revolution and beyond, Britain played a major role in world commerce and by the start of the First World War possessed over 40 % of world tonnage (McConville 1977). In the second half of the 20th century many dockland areas in Britain went into decline. This was a reflection of the trends in commerce and industry and the changing ship sizes and cargo handling practices. The larger ships coming into use could not ascend into the estuaries nor use small dock basins designed for sailing ships and small steamers. Consequently the first docks to suffer were those in the upper and middle reaches of estuaries. The increasing shift to containerized cargoes required the development of new terminals at the seaward edge of estuaries with large landward storage areas and easy access to trunk roads (Church 1988, Hayuth 1988, Hoyle 1988). Switching of cargoes from small coasters to road haulage also diminished demand at many docks (Johnson & Garnett 1971). Thus from the 1970's onwards large tracts of dockland in many inner city areas became disused and derelict.

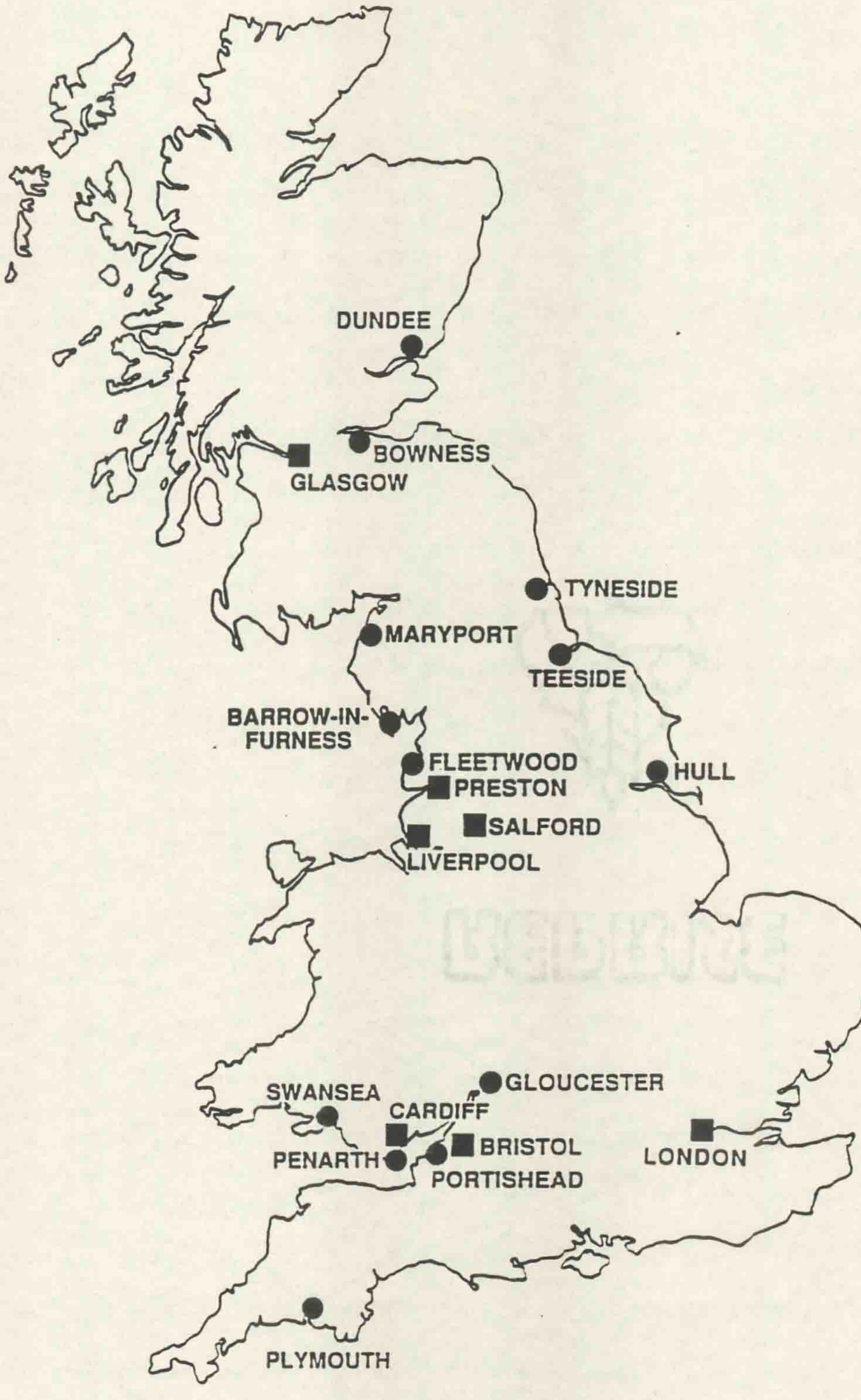
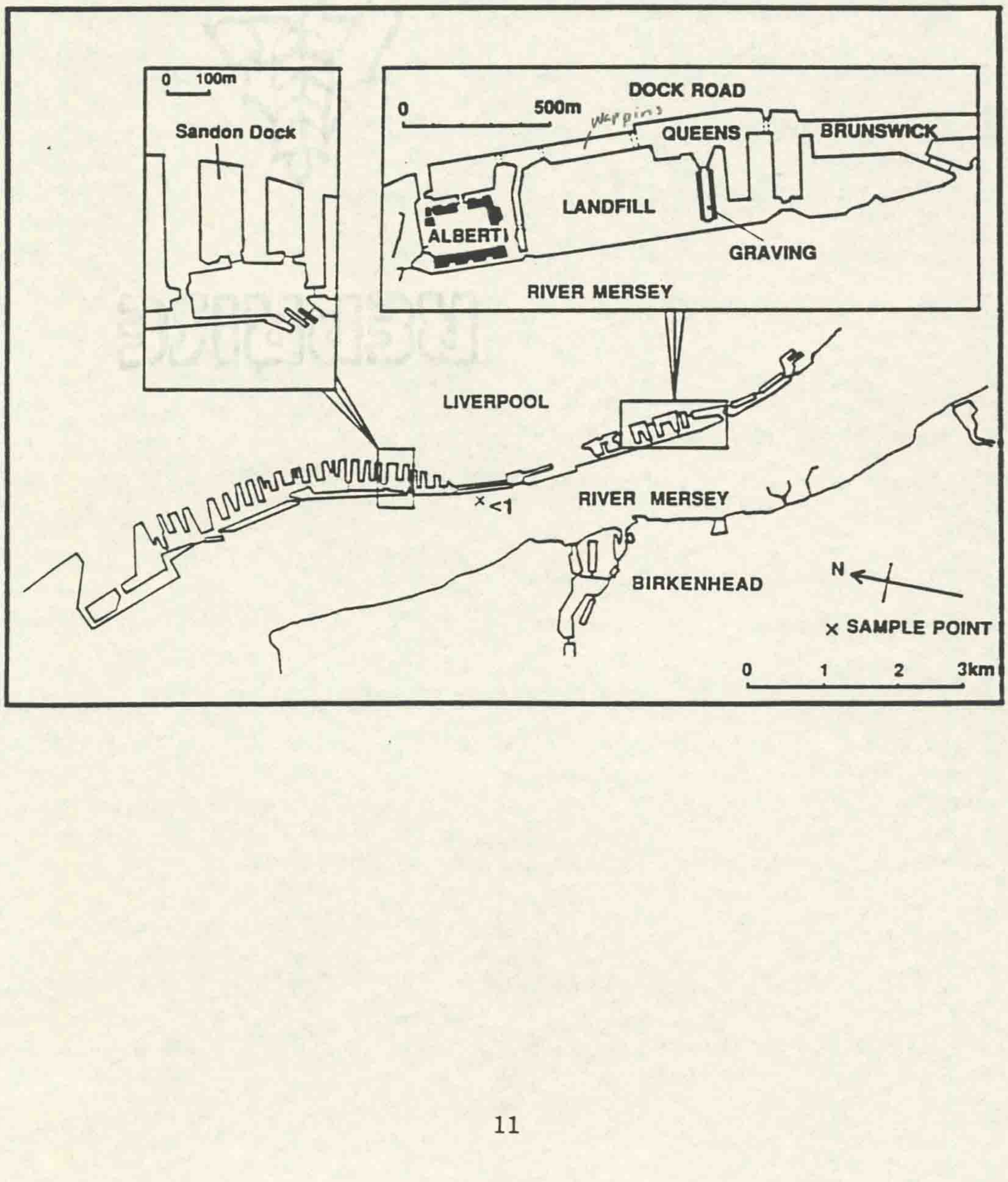


Fig. 1.1 Disused docks in the U.K. Squares indicate major redevelopment schemes, (after Hawkins *et al* in press b)

Fig. 1.2 Location of Sandon Dock and the South Docks, Liverpool, (after Hawkins *et al* in press b).



British docklands have been the focus of various inner city regeneration projects in the 1980's. These usually involve some combination of housing (often luxury), retail, business, light industry, cultural and recreational developments. These developments are centred around the appeal of an attractive waterside location. Dock basins have also been used occasionally for aquaculture or scientific research and there has been a growing realization of the role that dock habitats can play in urban nature conservation (Russell *et al* 1983, Cunningham *et al* 1984, Hendry *et al* 1988 a, b). All such redevelopments are dependent on good water quality. Unfortunately many dock basins draw their water from industrialised estuaries and are consequently badly polluted, resulting in unsightly water bodies.

In this thesis I concentrate on the hydrography, ecology and water quality management of high salinity disused docks, using the South Docks in Liverpool as an experimental system. To place this work in context, in the remainder of this introduction, the history of the South Docks and existing knowledge on the ecology of the Mersey Estuary are briefly described, the findings of previous studies on the water quality and ecology of other dock complexes in the U.K. are then reviewed, along with comparable work from other coastal enclosed water bodies such as lagoons and embayments. The developments in methods of water quality control which have taken place over the last 30 years are then summarized. Finally the specific aims of this research project are outlined.

1.2 HISTORY OF THE SOUTH DOCKS

The port of Liverpool was well situated for the growing trade to Africa and the Americas at the end of the 17th century. The strong tides, large tidal range, strong winds and shifting mudbanks of the Mersey, however, presented a major restriction to the growth of shipping trade. In response to this pressure the world's first commercial maritime dock was constructed within the confines of a shallow creek (known as 'The Pool') which had previously provided some shelter for ships. This Dock (Old Dock) retained water at all states of the tide and allowed unrestricted loading and unloading of ships directly into warehouses on the quayside. The construction of this dock sparked off a period of further growth in trade centred

Plate 1.1 Albert Dock in 1982, prior to dredging and restoration.

Plate 1.2 The South Docks, from Wapping to Canning Docks in 1988 (three to four years after refilling with water).

on Liverpool which was sustained by the continual construction of new docks. The topography of the area and methods of construction dictated that the docks expanded in a narrow ribbon along the Mersey. At its peak Liverpool boasted over 100 docks stretching from the river mouth to 10 km upstream.

The South Docks (Fig 1.2) contain some of the oldest remaining docks on Merseyside, dating back to the mid 18th century. The dock complex gradually evolved, with the last major construction in the South Docks taking place at the beginning of the 20th century. Since then the docks have consisted of a chain of docks with only two river entrances, one at the northern and one at the southern end (Ritchie-Noakes 1984). The decline of the South Docks began after the Second World War and accelerated in the 1960's. In 1972 the docks were closed to commercial shipping. Subsequently the gates of the docks were left open and silt deposited by the tides quickly built up. Within 10 years the mud was 10m deep in places and was only covered by water at high tide (Plate 1.1). The immense brick warehouses dating back to the mid 19th century fell into dereliction.

This state of affairs came to an end in 1981 with the setting up of the Merseyside Development Corporation, the aims of which were to develop the docks as a commercial project with funding from both public and private sectors. Dredging of the docks began in 1981, starting at Canning and Albert Docks and working southwards to Brunswick Dock. Water was gradually replaced as dredging progressed, finishing in 1985 with replacement of the double lock gates at the Brunswick river entrance. Details of the size, depth and physical characteristics of the South Docks are given in Chapter 2.

A variety of schemes have now been completed. The historic buildings of the Albert Dock now house offices, luxury flats, a major art gallery, the Maritime Museum and retail outlets, with further accommodation at Wapping and Coburg Docks. The Albert Dock is a major tourist attraction receiving over 3.5 million visitors per year. Watersports are centred on Queens Dock and there is a rapidly expanding marina in Coburg and Brunswick Docks. Developments

are continuing with the construction of a new Customs and Excise building which will span the Graving Dock and further recreational and housing projects are planned. As yet plans for an aquarium have not been realised but are under discussion again at the time of writing.

1.3 ECOLOGY AND WATER QUALITY OF THE MERSEY ESTUARY

The source of water for the South Docks is the Mersey Estuary which is reputed to be one of the most polluted estuaries in Britain (Clark 1989). The estuary is flanked on both sides by industrial plants which continue along the river corridor and Manchester Ship Canal. The predominance of chemical and petrochemical industries has led to several pollution incidents, such as the death of waders and gulls from lead pollution (Head 1980), and the recent oil spill from a fractured pipe line (Hall-Spencer 1989). Industrial discharges in the Mersey have also led to chronic contamination of biota and sediments with metals and other persistent chemicals, evidence of which is seen some distance out into Liverpool Bay (Dickson & Boelens 1988, Irish Sea Study Group 1990). Discharges from modern industrial chemical plants often contain a complex assemblage of compounds, many of which cannot be identified using standard techniques and for which no ecotoxological studies have been carried out (Greenpeace 1990, Foundation for Water Research 1990). Liverpool Bay is reported to have the highest levels of persistent synthetic organic compounds in the Irish Sea (Irish Sea Study Group 1990). Levels of the pesticide Lindane have also been found in water which are above the safe levels recommended by the E.C. (Irish Sea Study Group 1990).

The high degree of organic pollution and associated low oxygen concentrations historically restricted biota in the Mersey Estuary. The Mersey Basin has a population of six million people, much of which is served by inadequate sewage treatment systems. This, combined with agricultural and industrial discharges has led to over half the river length in the Mersey Basin being classified as Grade 3 or 4 (poor or bad water quality , Mersey Basin Campaign (MBC) public information literature). Not surprisingly, levels of sewage derived micro-organisms are high in the lower estuary (National Rivers Authority (NRA) data 1989, see Table 2.2).

The problems of pollution in the Mersey Estuary have been decreasing since the mid 1960's (Wilson *et al* 1988, Irish Sea Study Group 1990). The clean up has been given extra impetus since the launch of the Mersey Basin Campaign in 1985. This project, with a budget of £ 4 billion from public, private and European Community (EC) funds, aims to promote all watercourses to at least Grade 2 ('fair') by 2010 (MBC public information literature 1990). One major initiative, a new sewage treatment works on the site of Sandon Dock, is now in operation.

Despite the pollution that is present in the estuary, it is far from lifeless. Recent studies on the fish of the estuary found a total of at least forty species, ten of which were fresh water species (Wilson *et al* 1988, Lonsdale 1990). Species diversity in samples was generally low with a dominance of sprats (*Sprattus sprattus*), juvenile herring (*Clupea harengus*) and sand gobies (*Pomatoschistus microps*). Good historical data exist for the benthos of the Mersey estuary which is reviewed in Mills (1991). An extensive study of the intertidal fauna was carried out by Bassindale (1938). Common species found in this study were *Hediste diversicolor* (also known as *Nereis diversicolor*) , the oligochaete *Clitellio arenius*, *Corophium volutator*, *Macoma balthica*, *Arenicola marina* and *Pygospio elegans*. A later study (Bamber 1988) reported a slightly more impoverished fauna dominated by oligochaetes in the upper estuary with *Tubifex costatus*, *Tubificoides benedini*, *Nephtys cirrosa* and *Capitella capitata* dominating at other sites. Pollution was identified as the most likely cause of these changes although a severely cold winter was cited as another possible factor. The biota of floating structures in Liverpool Bay and Estuary were described by Fraser (1938) and Corlett (1948). *Mytilus edulis* was found to be the dominant species in both the estuary and inner bay. Hydroids (*Tubularia* spp. and *Laomedea* spp.) and several species of balanoid barnacles were also common. Occasional blanket settlement of *Polydora ciliata* was recorded in the lower estuarine sites.

Little published information is available recently on the plankton of the Mersey Estuary. What has been done is mostly centred around studies of the effects of sewage sludge dumping or the proposed barrage scheme (Sharples 1972, Mersey Barrage Company internal reports 1991). The phytoplankton of the Mersey is characterised by a *Skeletonema costatum* bloom in early spring, followed by dense populations of *Phaeocystis pouchetii* and large numbers of the dinoflagellate *Noctiluca scintillans* in summer months (Sharples 1972, Burrows 1975, G. Russell unpub., A.Jemmett pers. comm.). *Eurytemora affinis* is the dominant copepod in the upper estuary and overlaps with an assemblage of various *Acartia* species (*bifilosa*, *discaudata*, *clausii* and *longeremis*) in the mid section and *Temora longicornis* and *Centropages hamatus* at the river mouth(Williamson 1975, Environmental Resources Ltd (ERL) data 1990). The most common larger predators are *Pleurobrachia pileus*, *Sagitta setosa*, *Aurelia aurita* and the mysid *Neomysis integer* (Pierce 1941, Williamson 1975, Dempsey 1987, ERL data 1990).

Over the last four years greater attention has been focused on the ecology of the Mersey Estuary because of the proposal for the construction of a barrage from New Ferry to Otterspool. Concern about the effects of this scheme on the environment and potential problems such as dinoflagellate blooms have prompted an intensive environmental survey (work carried out by ERL Ltd on behalf of the Mersey Barrage Company). It is my opinion that the study of the South Docks, which are located within 2 km of the proposed barrage site, should provide some insight into the effects of impounding Mersey Estuary water.

1.4 PREVIOUS WORK ON DOCKS

Since the early 1980's research on the water quality and ecology of disused docks has been increasing, prompted by the redevelopment of many docks over this period. Much of the work has therefore centred around the major developments at Liverpool, Salford and Preston Docks.

A study of the sediment benthos and heavy metal contamination in the polyhaline Collingwood Dock, Liverpool, was carried out by James & Gibson (1980). Sediment fauna was found to be dominated by *Capitella capitata* and levels of lead and zinc were higher than in most estuaries. Work at another Merseyside dock, Sandon Dock, was centred around the experimental use of the dock for aquaculture which involved investigations into mussel growth and settlement, water quality, benthic ecology and plankton (Russell *et al* 1983, Naylor 1983, Cunningham *et al* 1984, Liebeschutz 1985, Hawkins *et al* in press a, b, c). Water quality in this dock was managed with an artificial mixer which prevented anoxic bottom waters. Increased water clarity was attributed to filtration of the water by cultured mussels. A surprisingly diverse flora and fauna developed with time. Heavy metal levels in mussel tissue were low in comparison to many industrialised estuaries.

Several short term studies on the water quality and ecology of the South Docks have been carried out during this research project. Several of these studies examined fish populations (Mincher 1988, Heaps 1988, Lonsdale 1990) which were dominated by whiting, flounder and sprat. Analysis of heavy metals in fish tissue revealed levels of lead which were above E.C. guidelines in whiting (Kreuser 1988).

The water quality, plankton, benthos and fish of Preston Dock, a low salinity dock, were studied over 28 months by Conlan (Conlan *et al* 1988, Conlan 1989). The water in this dock was of poor quality, suffering from periods of anoxia, often in association with a halocline. Coliform bacteria levels were frequently high and dense blue green algal blooms occurred throughout most of the year. No easy solution could be found for the problems of this dock.

The large complex of freshwater docks at Salford, Greater Manchester, are now in the later stages of redevelopment and have been the subject of long term studies going back to before redevelopment (Montgomery 1987, Boyd 1989). 'Helixor' water mixers are used continuously in the redeveloped basins and have eliminated thermal stratification and low oxygen concentrations (Radway *et al* 1988, Hendry *et al* 1988a). Mixers have not been successful in controlling blue green algal blooms however, and dense populations are present throughout the year (Hendry *et al* in press, Bellinger *et al* in press). Some of the dock basins at Salford Quays are now being developed as a recreational fishery for coarse fish (White *et al* in press). The potential of a variety of docks as a habitat for natural populations of freshwater and marine fish has also been considered (Conlan *et al* 1988, Hendry *et al* 1988b).

The urban location of docks, often presents problems when they are used for water sports. This is highlighted by Enticott & Grisdale (1985) for Bristol Docks, which received direct discharges of sewage despite being used for recreational purposes. In these docks several people suffered from gastroenteritis after a watersports event, and one person died after contracting Weils disease, despite the docks being within EC bacteriological guidelines at the time.

In a broad survey of 10 docks throughout the U.K. Hendry *et al* (1988a) found that problems of high nutrient levels, algal blooms, anoxic water and poor species diversity were common to most docks. These problems are essentially those of eutrophication and seem to be typical of docks in general. The eutrophic conditions stem from the poor quality of waters with which the docks are filled. A comprehensive nationwide appraisal of U.K. docks, their ecology, water quality problems and possible solutions is given in Hawkins *et al* (in press, b) and their value for urban nature conservation and use in education is considered in Hawkins *et al* (in press c)

1.4 COMPARABLE HABITATS: COASTAL LAGOONS, ENCLOSED ESTUARIES AND SALINE LAKES

The South Docks are an enclosed, shallow, brackish water habitat with very restricted external water exchange. Comparable man-made and natural habitats are found world-wide. Previous studies of these habitats may provide an insight into the functioning of the dock ecosystem.

Brackish water coastal lagoons are a similar habitat which may be created by natural topography or as a result of mans activities. They are shallow areas of water separated from the sea by a narrow land barrier. Water influx may be by percolation or through an inlet channel. Coastal lagoons are rare in Britain and in Europe as a whole. Europe has the lowest proportion of lagoonal coastline of any continent at 5.3% (Barnes 1980). The ecology and water quality of lagoons has been studied in several British lagoons (e.g. Hill *et al* 1987, Barnes 1980, 1987, 1988a, Crawford 1978, Dorey *et al* 1973), as well as in other lagoons worldwide (e.g. Millan-Nunez *et al* 1982, Nixon 1982, Grizzle 1984). An overview of the ecology of coastal lagoons in Britain was given by Barnes (1988b), in which 38 specialist lagoonal species were identified. A biological survey of over two hundred British saline lagoons was carried out by the Nature Conservancy Council (NCC) between 1984 and 1989. Smith & Laffoley (1992) outline the biological and geophysical status of the English lagoons surveyed as part of this project. Only a small proportion of the coastal waterbodies surveyed were 'natural lagoons' however, having a total area of 660 hectares. Other brackish, coastal water bodies included boating lakes and drainage ditches.

Land reclamation, port expansion and coastal protection works have all contributed to a reduction in the number and extent of lagoons (Barnes 1991, Hawkins *et al* in press b). It has been suggested that man-made lagoons could possibly be used to preserve specialised lagoonal species (Barnes 1991). Disused docks may help to provide suitable habitats for lagoonal organisms (Hawkins *et al* in press b).

Estuaries and bays with restricted entrances such as fjords and sea lochs may also have features in common with disused docks (e.g. Winter *et al* 1975, Jørgenson 1980) although hydrodynamic residence times are generally shorter and tidal currents may occur.

Of other man-made systems, saline lakes, such as those created along the Dutch coast often have the most similar physical characteristics to disused docks, such as shallow depths, long hydrodynamic residence times, brackish water and elevated nutrient levels. The Dutch saline lakes have been the subject of detailed ecological study over the last 20 years (e.g., Bakker & Pauw 1974, Nienhuis 1978, Vries & Hopstaken 1984, Lambeck & Valentija 1987, Rijstenbil 1987).

It is perhaps not surprising that lagoons and other enclosed brackish waterbodies often suffer from similar water quality problems to disused Docks. These habitats often show signs of eutrophication and dense phytoplankton blooms, sometimes of toxic dinoflagellates and cyanobacteria are reported (e.g Crawford *et al* 1979, Jones *et al* 1982, Silva 1985). Salinity or thermal stratification of the water is often seen and this, possibly combined with high organic loading leads to low dissolved oxygen concentrations in deeper layers (e.g. Dorey *et al* 1973, Crawford*et al* 1979, Grizzle 1984, Jorgensen 1980).

1.6 WATER QUALITY AND MANAGEMENT OF ENCLOSED WATER BODIES

The water quality problems of most aquatic bodies are associated with high nutrient loading leading to eutrophication, for example, low hypolimnetic oxygen concentrations and problem algal blooms. The only long term solution to eutrophication is the reduction of nutrient inputs (e.g.Edmonson 1970, Laurent 1971). The implementation and action of such strategic approaches may take many years, or may be impractical to some extent. Fast acting methods are available which deal with the symptoms of eutrophication. Almost all of these methods have been developed, and are now used, in freshwater lakes. This is due partly to their

proximity to urban and agricultural areas which makes lakes prone to eutrophication, but also prompts great pressure for a rapid solution to the aesthetic and public health problems. Also the small size of many freshwater lakes makes management feasible.

Direct chemical techniques for reducing algal biomass have been used with some success. Chemical strippers, such as aluminium sulphate can be used to flocculate and settle out nutrients (Wall 1971, Cooke & Kennedy 1980, Kennedy & Cooke 1980, Soltero *et al* 1981). The application of algicides such as copper sulphate has also been used (Landner 1976). Such applications are expensive and the beneficial effects are short lived. Detrimental effects may also be seen higher up the food web.

Since the early 1960's artificial mixing of water has been used in lakes and reservoirs to increase hypolimnetic oxygen concentrations and hence combat immediate problems of fish kills and tainted water supplies (e.g.Knoppert *et al* 1970, Fast 1973, Bailey-Watts *et al* 1987). Most water mixers work by the release of air bubbles into the hypolimnion, which entrain water as they rise and increase circulation, although direct movement of water by paddles or jets may be used (see reviews in Tolland 1977, Pastorok *et al* 1981). It is often presumed that, in deep lakes, mixing will decrease phytoplankton biomass by light limitation as algae are able to spend less time in the photic zone (Pastorok *et al* 1980). The effects of mixing on phytoplankton are somewhat unpredictable however, a review of such effects is given by Pastorok *et al* (1980). Water mixers have been rarely used in enclosed high salinity water bodies, partly due to the scarcity of suitably sized problem areas. One example of such use was in Sandon Dock, where mixing was used to raise oxygen concentrations to levels compatible with use for aquaculture (Russell *et al* 1983).

Management of algal biomass by artificial mixing or by nutrient reductions is an example of "bottom-up" control. Mixing may have direct inhibitory effects on phytoplankton by increasing light limitation by reducing residence times in the photic zone. Mixing may also reduce the supply of nutrients for phytoplankton growth by lowering the rate of release from

the sediments. The manipulation of higher trophic levels provides a "top-down" approach to phytoplankton control by raising levels of grazing (Gophen 1990). The deliberate exploitation of the interactions between the components of the aquatic ecosystem in order to reduce algal blooms was termed biomanipulation by Shapiro *et al* (1982). The development of biomanipulation techniques and a review of current knowledge is provided by Lammens *et al* (1990). Biomanipulation has been restricted almost entirely to freshwaters. Successful manipulations producing top-down effects have mostly concentrated on the ultimate control of phytoplankton by large-bodied zooplankton. Various methods have been used to boost zooplankton populations and hence reduce phytoplankton biomass. A reduction in numbers of planktivorous fish by direct removal or stocking with piscivores has been used (e.g. Lynch & Shapiro 1981, Dorazio *et al* 1987, Bendorff 1990, Donk *et al* 1990, Riemann *et al* 1990, Sanni & Waervågen 1990). The provision of refuges, which may work by protecting zooplankton from predators, or by reducing predator efficiency, has proved effective in some cases (Moss 1990, Shapiro 1990, in press). Such refuges may be provided by: low light intensity, low temperature, low dissolved oxygen, physical concealment, visual clutter, modification of predator behaviour or predator inefficiency mechanisms (Shapiro 1990). The future use of the freshwater benthic filter feeding bivalve *Dreissena polymorpha* has also been proposed, populations of which can be increased by providing suitable settlement substrates (Reeders 1989, Reeders & Bij de Vaate 1990).

Improvements in water clarity due to reduced silt suspension and "bottom-up" control of phytoplankton is also possible by removal of benthivorous fish leading to reduced benthic bioturbation (e.g. Tatrai *et al* 1990, Meijer *et al* 1990). The harvesting of macrophytes as a form of nutrient removal has also been attempted and may be useful in lakes in which nutrient input is already controlled (King & Burton 1980), although this practice may reduce zooplankton populations and hence stimulate phytoplankton blooms (Lembi *et al* 1978).

In saline systems biomanipulation has not been developed, the size and openness of most marine systems making this impractical. The influence of natural top-down effects of benthic filter feeders has been observed in some shallow, partially enclosed coastal and estuarine areas (e.g. Officer *et al* 1982, Davies *et al* 1989, Hily 1991).

The effects of biomanipulation may be short-lived and continued management of the aquatic ecosystem is often required to maintain results (Lammens *et al* 1990, Shapiro 1990). The stability of the effects of biomanipulation may be improved by concurrent reductions in nutrient inputs (Scheffer 1990), by using several different species for the biomanipulation (Benndorf 1990) and in the case of zooplankton, by providing refuges as a protection from predation (Moss 1990, Shapiro 1990). The most positive results of biomanipulation in freshwaters have been achieved in shallow waters with small volumes and over short time scales. Gophen (1990) emphasises that to increase volume, depth and time scales of future manipulated water bodies an integrated and multi-disciplinary approach is required.

The attempts at water quality control carried out during my research project were stimulated by observations made at Sandon Dock (Russell *et al* 1983, Cunningham *et al* 1984). Improvements in water quality in terms of oxygen concentrations and clarity were observed in this dock when an artificial mixer was used and large mussel population was present in the dock as part of aquaculture operations. The improvements in water clarity were attributed to filtration of the water by cultured mussels.

1.7 AIMS OF THIS RESEARCH PROJECT

This project was funded by an urban development agency, the Merseyside Development Corporation (MDC), initially on a 12 month contract, with renewal occurring on a year by year basis after this time for a further 21 months. Long term planning of research was therefore very difficult. Hence the sampling and experimental programme could have been better

planned in several aspects, if the duration of the project had been known from the outset. The source of funding has also meant that this research is of a very applied nature. The major aims of the research were:

1) To gain an understanding of the overall hydrography and ecology of the South Docks complex and, particularly, to identify the extent of water quality problems.

2) To develop a management strategy for dealing with any water quality problems that might occur and for maintaining high water quality in the South Docks in the longer term.

Chapter 2 is an introductory chapter giving background information on the dimensions of the docks, locations of sampling sites and background physical information relevant to all of the subsequent chapters.

Chapters 3 to 5 are concerned with the hydrography and ecology of the docks and are of a descriptive nature. Chapter 3 describes the physical and chemical nature of the dock environment with attention to annual and longer term variation and identifies any associated water quality problems. Chapter 4 details temporal and spatial variations in plankton biomass and species composition, again related to water quality. Chapter 5 is a very preliminary study of the benthic and nektonic communities, based in some cases on incidental observations. In this chapter most attention is given to the distribution and population dynamics of the natural mussel populations, because of their possible role in water quality management.

The second research aim, that of developing management strategies, is of a more experimental nature and is covered in chapter 6. This chapter describes the effects of an artificial water mixer and an introduced population of filter feeders (*Mytilus edulis* L) on the water quality of a semi-isolated experimental dock. The effects of a natural population of mussels, which

settled during the research project, on phytoplankton biomass is discussed. The recommendations for future water quality management, as given to the MDC, are also included.

Chapter 7 is a general discussion of all the work. It considers shortfalls in methodology, the functioning of the South Docks ecosystem, and the possible application of the management techniques used to other systems. A brief appraisal of the conservation value of disused docks in general and the South Docks in particular is also made, along with recommendations for further work.

CHAPTER TWO

PHYSICAL AND ENVIRONMENTAL BACKGROUND

2.1 INTRODUCTION

In this chapter the South Docks are described as they are today in terms of lay-out, size and operation. The position of the main sampling sites are given. Finally, background information, relevant to more than one of the later chapters, is given.

2.2 SIZE AND LAY-OUT OF THE SOUTH DOCKS

The South Docks now consist of a chain of dock basins, stretching for approximately 2 km along the eastern bank of the River Mersey from the city centre, southwards upstream. Two sets of gates connect the docks with the river, one at Brunswick Dock to the south, the other at Canning Dock to the north. Canning river entrance is a single gate connecting the dock directly with the river, another gate then joins Canning with the Albert Dock and the rest of the dock system. Beyond Albert Dock, a chain of connected dock basins leads southwards to Brunswick Dock and the second river entrance (Fig. 2.1). This consists of double gates with a large lock space which allows several pleasure craft to be locked in or out at one time. Brunswick Lock is the main point of passage for boats to and from the South Docks, with Canning entrance used mainly for exhibition craft and tall ships. Locking in and out of Brunswick lock can only be carried out for two hours either side of high tide.

The total area of waterspace from Brunswick to Canning Docks is 29.8 hectares, with a water volume of around 15.2×10^5 m^3. Areas and volumes of individual dock basins are given in table 2.1, along with the areas of other redeveloped docks in Britain for comparison. Dock areas for the South Docks were calculated from draughtsmans maps. Volumes were calculated using average depths for each dock estimated from sonar surveys.

The majority of dock basins in the South Docks are dredged to between 3 and 4 m depth. Albert Dock is dredged to 6 to 7 m and the Graving Dock is 10 m deep. The docks typically consist of sheer stone or concrete walls with a thick layer of fine sediment on the bottom. The Graving Dock retains the stepped walls typical of this type of dry dock.

2.3 SAMPLING SITES

The main sampling points for water and plankton sampling were located in Graving, Albert and Queens Docks (see Fig. 2.1 and Plates 2.1 and 2.2). The Graving Dock was chosen as a semi-isolated experimental dock for the introduction of mussels and mixing trials. It is

Table 2.1 Approximate areas, maximum depth and volumes of docks within the South Docks Complex. Areas of other redeveloped docks are given for comparison (from Hawkins *et al*, in press b).

Dock	Area (m2 x 10^4)	Approx. max depth (m)	Volume (m^3x 10^5)
Albert	2.6	7	1.6
Salthouse	2.6	4	0.9
Dukes	0.6	4	0.2
Wapping	2.3	4	0.9
Queens (incl. branch dock)	6.9	4	2.8
Graving	0.5	10	0.5
Coburg	3.0	4	1.1
Brunswick	6.6	4	2.6
Canning (incl.Half Tide)	2.7	5	1.3
Total water space Albert to Brunswick including passages.	29.8		11.9
Sandon Dock	4.3	10	
Salford Quays	11.3	9	
Preston Dock	16.2	8	
London (Royal)	89.3	11	

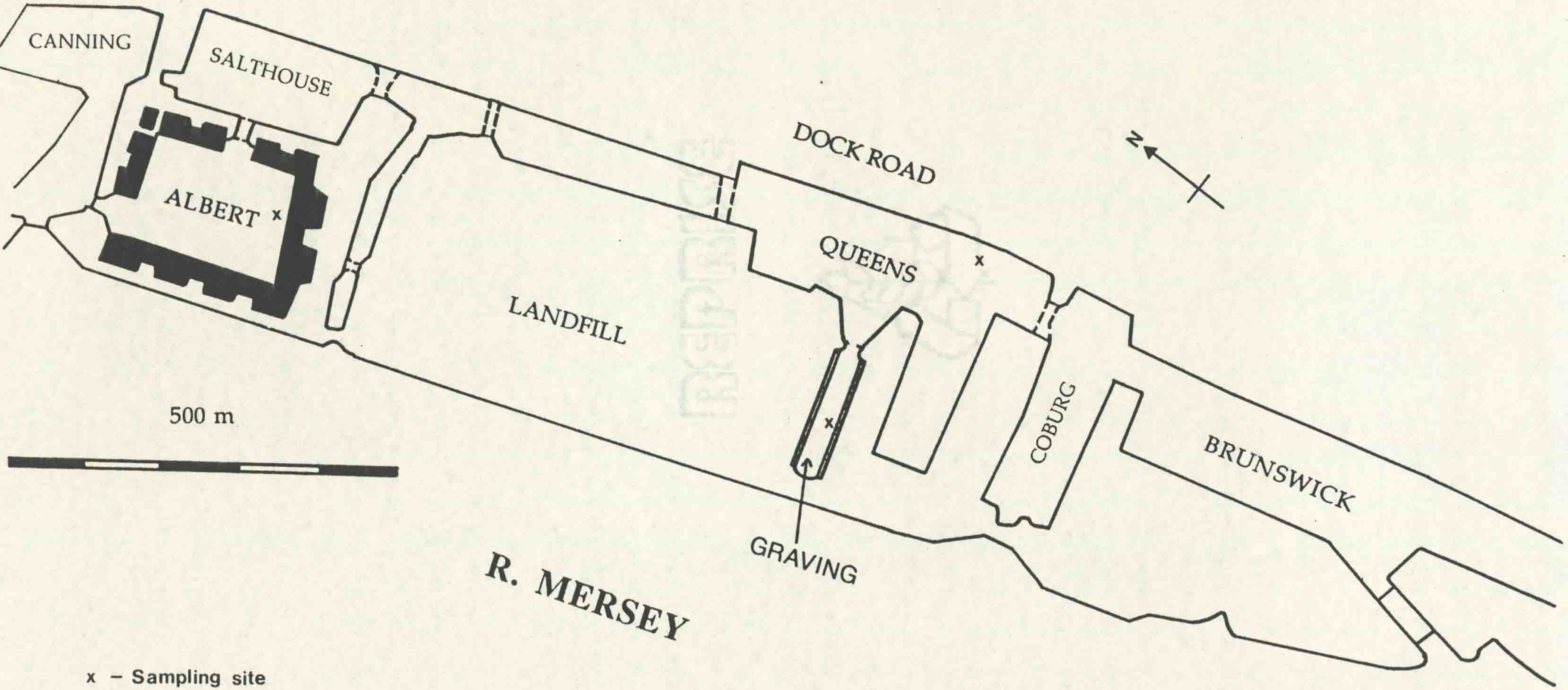


Fig 2.1 The South Docks Liverpool, Canning to Brunswick docks, showing position of sampling sites.

separated from the other docks by wooden gates. A small gap between these gates allowed the water to equilibrate between the two sides. A Martec Systems 'Rotamixer' air lift water mixer was placed in this dock at the beginning of the research project. Albert Dock was initially chosen as a control dock without mussels, being the nearest in depth to the Graving Dock, although very different in shape and size. Unfortunately a natural settlement of mussels in the Albert Dock in 1988 thwarted plans to use it as a control. The Queens Dock was chosen as being typical of the majority of the South Docks being three to four metres deep.

Sampling from permanently moored pontoons allowed water samples to be taken away from the dock wall in the Albert and Queens Docks, but in the Graving Dock a raft was required to allow sampling away from the stepped walls.

2.4 WATER INTAKE

Water is lost from the docks by seepage, evaporation, locking operations and occasional flushing through sluice gates to keep the approaches free from silt. Water is replaced by exchange with the Mersey Estuary approximately once every two weeks, on spring tides at high water. Water is taken in through sluice gates at Brunswick Lock. Between 0.1 and 0.5m depth of water is normally taken in at each fill. This is approximately 3 % to 13 % of the dock volume from Brunswick to Albert, water does not penetrate into Canning Dock which is separated by gates and is operated as a separate system. Inadvertent inputs of water to the Albert Dock system from Canning Dock will occur very occasionally when boats are admitted to Albert Dock from Canning Dock at a time when the water level is lower in the Albert. The amount of water intake by this route is negligible, however, because boat traffic by this route is usually very infrequent and water levels are often lower in the Canning Dock than in Albert (MDC information) Further details of annual inputs over the sampling period are given in chapter six. The quality of the intake water is very poor: it is high in suspended solids, sewage derived micro-organisms and plant nutrients, and often containing low levels of dissolved oxygen (Table 2.2). As water intake to the dock is always carried out on spring tides at high water variation in the salinity of intake water will be relatively small.

2.5 WEATHER

The weather will have a bearing on both water quality and the general ecology of the dock system, the most important factors being air temperature, wind speed and levels of incident radiation. Observations from local meteorological stations were used to illustrate the conditions

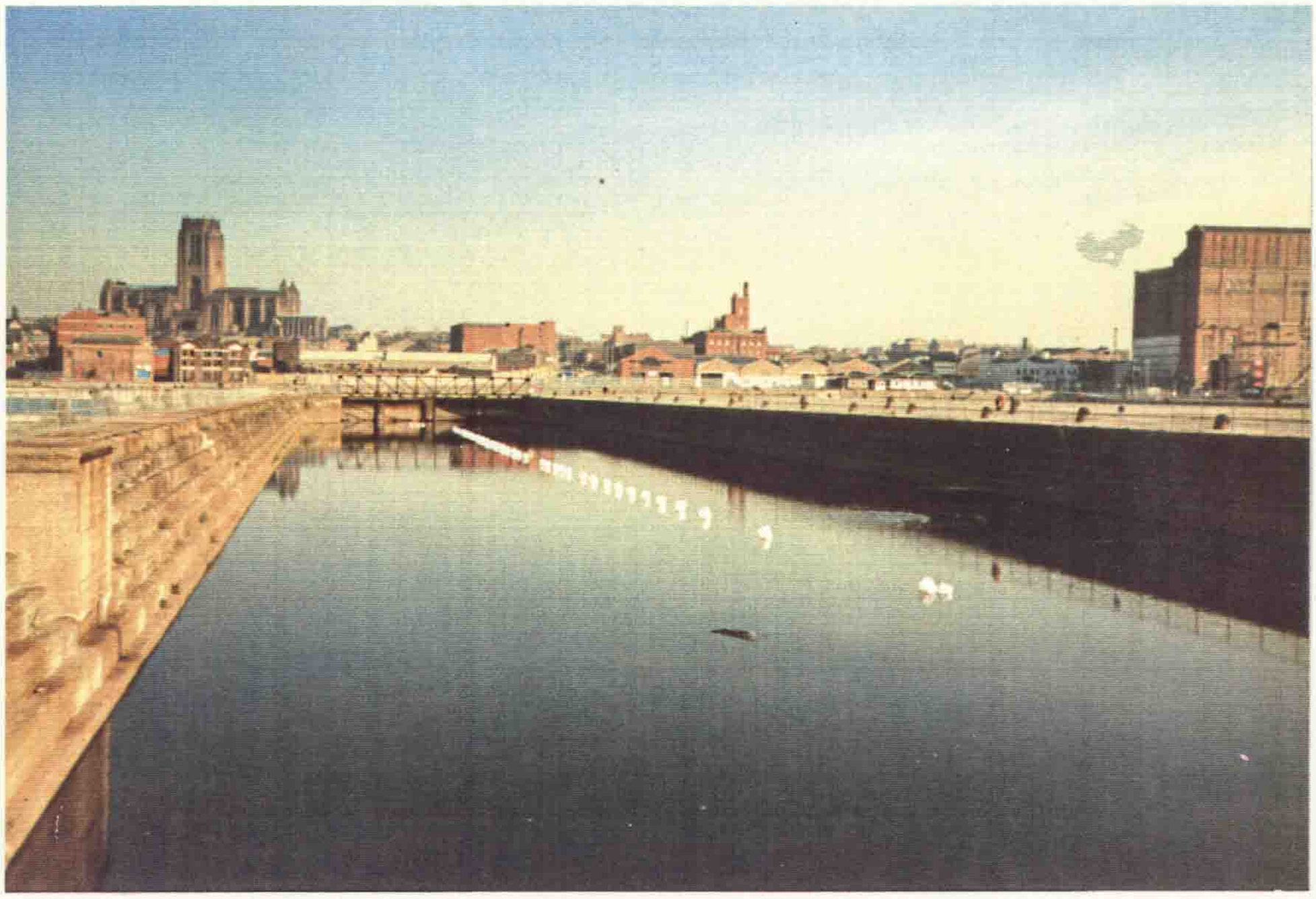


Plate 2.1 The Graving Dock (1989), showing the longline system for cultivation of mussels and the plume from the water mixer (centre right).

Plate 2.2 Albert Dock (1989), picture taken from sampling position, facing north.

from March to September of each sampling year (Fig 2.2). This is the time when changes in such parameters as wind speed and temperature will have the maximum effect on water quality and primary production, particularly by affecting the degree of stratification. Wind speeds and air temperatures were taken from the Aigburth meteorological station records, while hours of bright sunshine recorded at Bidston were used (Fig 2.2). Wind values for August 1988 and March 1989 were not available.

2.6 NOMENCLATURE

Throughout this thesis nomenclature follows Howson (1987) for fauna and South and Titley (1986) for macroflora unless otherwise stated. Species names for dinoflagellates are after Dodge (1982) and diatoms are as in Hendey (1974).

Table 2.2 Nutrients, dissolved oxygen, total coliforms and suspended solids in water from the Mersey Estuary, close to the point of water intake for the South Docks. Data from National Rivers Authority (1989). Means and range of monthly values taken opposite Brunswick Dock gates at high tide.

	Mean	Range
Orthophosphate (mgl^{-1})	0.18	<0.05 - 0.54
Total inorganic nitrogen (mgl^{-1})	1.02	0.54 - 2.00
Dissolved oxygen (mgl^{-1})	6.3	1.4 - 9.2
Total coliforms ($100ml^{-1}$)	16200	2700 - 95000
Suspended solids (mgl^{-1})	67	21 - 178

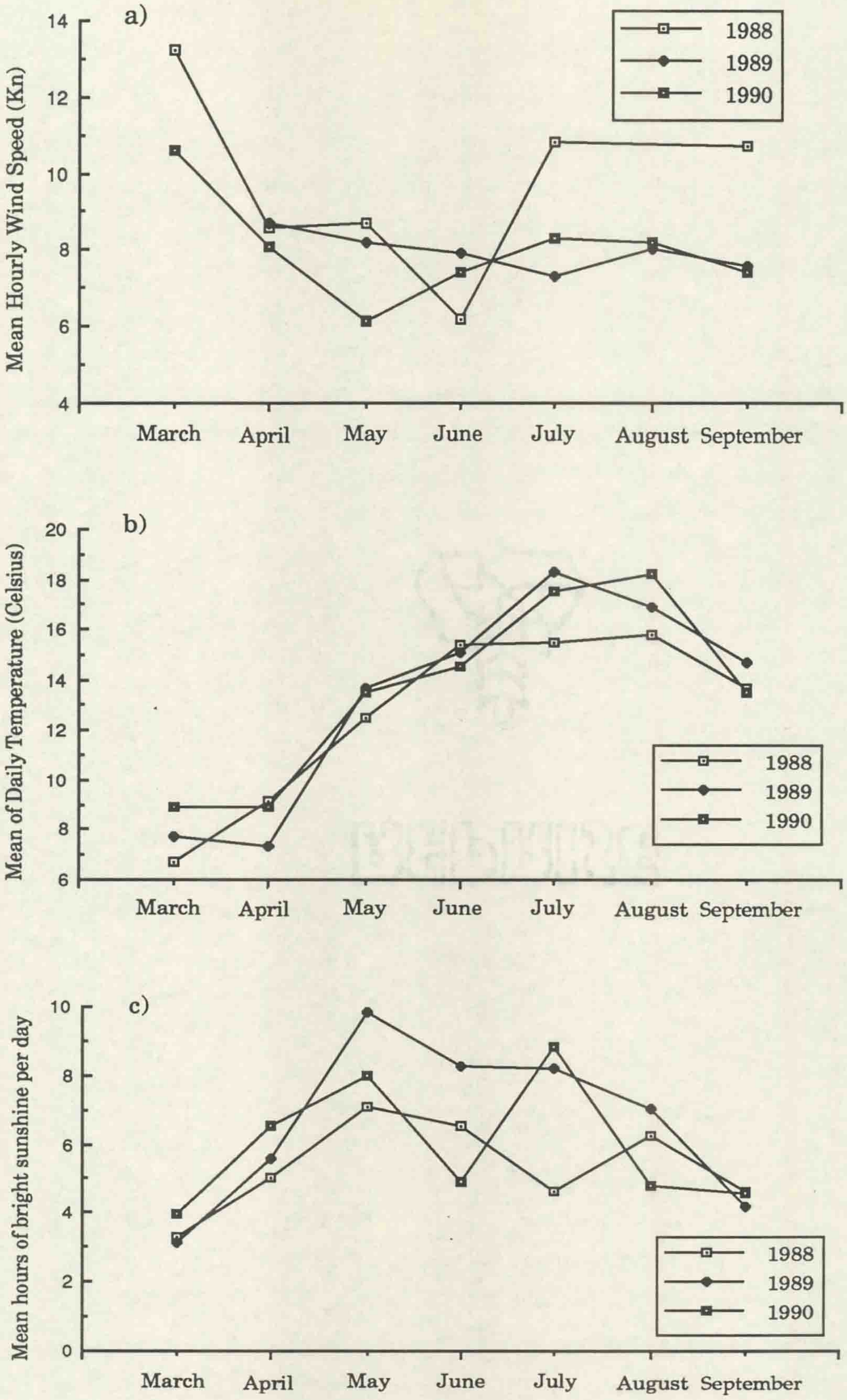


Fig 2.2 Weather data for summers 1988, 1989 & 1990: a) wind speed b) temperature, c) hours of bright sunshine. Data from local monthly weather reports, meteorological office.

SECTION ONE

DESCRIPTIVE ECOLOGY

CHAPTER THREE

TEMPORAL AND SPATIAL VARIATION IN PHYSICO-CHEMICAL PARAMETERS

3.1 INTRODUCTION

The ecology of disused docks and the extent of water quality problems depends largely on the hydrographic regime. Little information is available on the hydrography and ecology of docks and detailed long term studies are particularly scarce.

A description of a limited range of hydrographic parameters of another Merseyside dock, Sandon Dock, has been given by Russell *et al* (1983) . Thermal stratification with associated anoxic waters and large annual fluctuations in water temperature were thought to have limited the survival of some species in this dock Remedial measures, particularly installation of a Helixor mixing device, allowed a diverse community to flourish (see chapter one).

Conlan (1989) studied a wide range of parameters over a 28 month period in Preston Dock, an oligohaline, upper estuarine dock. In Preston Docks both salinity and thermal stratification resulted in depletion of hypolimnetic oxygen. Water clarity was poor throughout the year due to phytoplankton blooms and influxes of river water (Conlan *et al* in press).

Changes in water quality at Salford Quays, a redeveloped freshwater dock system, following isolation from the Manchester ship canal and installation of a mixing device are described by Radway *et al* (1988). Improvements in visual appearance and reductions in bacteria, organic matter nutrients and algal growth were attributed to this treatment, although problems of blue green algal blooms still remain (Hendry *et al* in press, Bellinger *et al* in press).

In a survey of ten U.K. docks carried out by Hendry *et al* (1988a) water quality was generally found to be poor due to the polluted waters which supplied the docks and poorly mixed conditions. High nutrient levels, thermal stratification, low oxygen concentrations and phytoplankton blooms were common water quality problems in these docks.

The South Docks, Liverpool, are in effect a series of semi-isolated brackish water lagoons. Detailed studies of water bodies with similar hydrographic properties have been carried out in several areas: Lake Grevelingen in the Netherlands is one such area. Grevelingen is a large, shallow, semi-isolated, man-made, brackish-water lake which has been the subject of detailed study since its creation in the late 1970's. Water quality is generally good, clear and mesotrophic (Bannink *et al* 1984). Primary production is limited by the supply of nitrogen and silica in summer (Bakker and De Vries 1984). Long-term observations indicate a slow eutrophication due to inputs from rainwater and polder run-off and decreasing denitrification (Bannink *et al* 1984, De Vries & Hopstaken 1984, Vegter & De Visscher 1984). Salinity stratification leading to oxygen depletion and benthic mortality was a problem in earlier years but this was reduced by the careful use of sluice gates. Other similar ecosystems (brackish water bodies with low water exchange) for which physico-chemical parameters are described include coastal embayments, lagoons, fjords and bays (Winter *et al* 1975, Taft *et al* 1980, Millan-Nunez *et al* 1982, Tenore *et al* 1982, Rijstenbil 1987, Paasche & Erga 1988)

This chapter is a study of the hydro-physical and hydro-chemical environment of the South Docks. This was considered essential to identify the nature and extent of any water quality problems and assist in the interpretation of the ecology of the South Docks. Although a three to five year period had elapsed since re-flooding, it was hoped to identify any trends following semi-isolation of the water body. Monitoring of physico-chemical parameters was also required for assessment of the success of remedial methods employed in the experimental Graving Dock. Key physico-chemical parameters were monitored at the South Docks over a 28 month period, which allowed assessment of any annual or longer term trends. Samples were taken at three sites with different characteristics (see chapter 2 for further details), to identify any spatial differences. Parameters studied were temperature, salinity, dissolved oxygen, water clarity, nitrate plus nitrite, ammonia, orthophosphate, reactive silicate, pH and biochemical oxygen demand.

3.2 METHODS AND MATERIALS

3.2.1 Field measurements

Field measurements, described below, were taken at the three sample sites from 26 th May 1988 to 12th September 1990. Measurements were carried out twice monthly from 26/5/88 to 28/11/88 and in the following summers and monthly during intervening winter periods. Missing results are due to instrument inoperation or with problems of access. Some additional records were made for specific experimental reasons or during periods of particular interest.

3.2.1.1 Salinity

From May 1988 to April 1990 an NBA Controls meter was used to measure conductivity (in m. mho/cm), at 1m depth intervals. Conductivity values were then converted to salinity in parts per thousand ($^{0}/_{00}$) using tables supplied by the manufacturer. This method allowed estimation of salinity to $\pm$ 1 $^{0}/_{00}$

In April 1990 the conductivity meter was damaged preventing its further use. After this time salinity was checked monthly on water samples collected for nutrient analysis using a 'Biomarine' refractometer. Salinity readings were taken to the nearest $^{0}/_{00}$. Accuracy of the instrument is given as $\pm$ 0.2 $^{0}/_{00}$.

3.2.1.2 Dissolved Oxygen And Temperature

Dissolved oxygen concentrations (mgl^{-1}) and temperature (Celsius) were measured using a pHox 62 combined oxygen and temperature meter. Instrument error was $\pm$ 1% for oxygen and $\pm$ 0.5 ^{o}C for temperature. Readings were taken at 1m depth intervals from just below the surface to approximately 0.5m above the sediment surface. Calibration of the oxygen meter was carried out according to manufacturer's instructions at the start of each sampling day. No temperature calibration was required but the meter was checked periodically against a mercury thermometer, instrument error was found to be within the stated range

on each occasion. Oxygen concentration values were used to calculate percentage oxygen saturation (H.M.S.O. 1979), using the appropriate correction factors for temperature and salinity.

3.2.1.3 Water Clarity

Water clarity was measured using a Secchi disc of 30cm diameter. The disc was lowered until no longer visible and then raised to the depth at which it just became visible. This was recorded as the Secchi extinction depth.

3.2.2 Laboratory Analysis

Water samples were collected and returned to the laboratory for determination of pH, biochemical oxygen demand (BOD) and plant nutrient concentrations. Samples just below the surface (0.2 m) were collected from each sampling site. In addition samples were taken from 5 m depth in the Albert Dock , and from 5 m and 9 m in the Graving Dock. Sub-surface samples were collected by hand, the inverted bottle being dipped below the surface, then righted and allowed to fill, thus avoiding the collection of any floating scum. An IOS remote-sampling bottle (1.5 l capacity) was used to collect samples at depth. Three replicate samples were taken at each depth, placed in clean 1l polyethylene bottles, covered and transferred the short distance to the laboratory as soon possible. All laboratory analyses were carried on a monthly basis from June 1988 to September 1990 unless otherwise stated.

3.2.2.1 Nutrients

Water samples for nutrient analysis were filtered, using a Buchner pump, through 7cm GF/C glass fibre filters, to remove particulate matter. Separate filter papers were used for each site, the initial 50 mls of filtrate being discarded in each case. Filtered samples were transferred to rinsed polyethylene bottles, iodised previously according to HMSO (1981) to reduce absorption of phosphate. Samples were stored on ice and analysed within 30 hours of collection. Concentrations of the plant nutrients orthophosphate, nitrate plus nitrite, ammonia and reactive silicate were determined in the three separate replicate samples from

each site/depth. A Skalar 5100 autoanalyser adjusted for low level determinations (detection limit 0.01 mgl^{-1}) was used for all analyses. Precision of all analyses is $\pm$ 0.01 mgl^{-1}.

Analysis of reactive silicate was not carried out until January 1990 when this facility was added to the autoanalyser. All other nutrients were measured on a monthly basis from June 1988 to September 1990 with an additional sample taken in January 1991 to assess winter levels for this period. The chemical basis of nutrient analyses on the Skalar autoanalyser are outlined below Methods are essentially the same as the for the manual procedures described in Parsons *et al* (1984) and automated techniques in Stephens (1970).

Orthophosphate was analysed by reaction with acidified ammonium molybdate solution to form a reduced phosphomolybdenum blue complex, which is measured spectrophotometrically. Interference with peaks was sometimes observed, particularly when orthophosphate levels were low. This is a common problem when measuring phosphate in saline waters (Crompton 1989). The recommended curative procedure of using saline blanks and standards did not greatly improve matters when phosphate levels were low, some loss of accuracy must therefore be accepted at these times.

Nitrate plus nitrite were analysed by dilution in ammonium chloride buffer and pumping through an activated cadmium column to convert nitrate to nitrite. A colour reagent was then added to form a red diazo complex with the nitrite ion which is measured spectrophotometrically. This method does not distinguish between nitrate and nitrite. Nitrite alone was measured on two occasions by omission of the cadmium reduction step. Levels were found to be very low. A maximum level of 0.02 mgl^{-1} was recorded from an anoxic water sample, but levels were generally below the detection limit of 0.01 mgl^{-1}.

The method of analysis of ammonia was by dilution in a sodium citrate and potassium sodium tartrate buffer to complex cations. A salicylate catalyst and activated chloride were then added to form a green coloured complex with the ammonium ion. The extinction was

measured at 660 nm and is proportional to the concentration of ammonia.

Reactive silicate concentration was determined by mixing with an acidic ammoniummolybdate solution to form silico-molybdic acid. Ascorbic acid was then added to reduce the silicomolybdic acid to a silicomolybdic blue complex. The extinction of this was measured at 720 nm, with oxalic acid being added to prevent interference from phosphate.

3.2.2.2 pH

pH of collected waters was measured immediately on returning to the laboratory, using a WPA pH meter (accurate to $\pm$ 0.1) calibrated on each sampling day with pH 7 and 9 buffers.

3.2.2.3 Biochemical Oxygen Demand (B.O.D.)

B.O.D. was determined monthly from 25/8/89 to 19/9/90 in water samples collected from each site and depth. The B.O.D. is an index of the amount of biologically oxidizable material available for depletion of oxygen in the water during degradation. B.O.D. is the amount of oxygen (in $mgl^{-1)}$ removed from a water sample when incubated in the dark for 5 days at 20°C. Oxygen concentrations were measured before and after the incubation period using a Kent EIL oxygen electrode, accurate to 0.1 mg^{-1}. B.O.D. was sufficiently low at all times that dilution of samples was not required.

3.3 RESULTS

3.3.1 Salinity

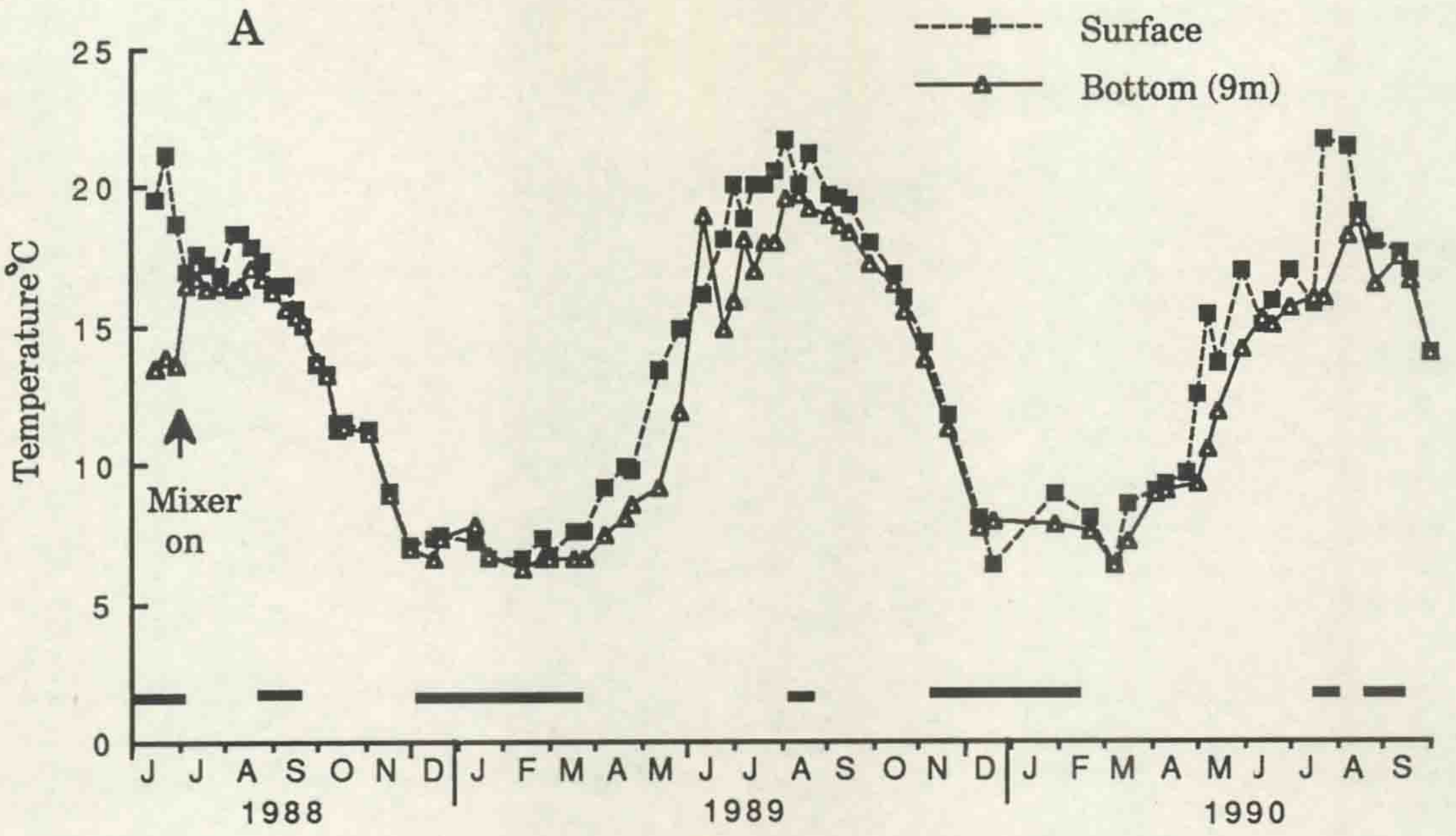
Salinity ranged from 23 °/oo to 28 °/oo in the Graving dock and 24 °/oo to 28 °/oo in the Albert and Queens Docks. No notable temporal or spatial trends were observed. Variation in salinity with depth was normally 1 °/oo or less from surface to bottom. A halocline was only observed on one occasion, in the Graving Dock, during December 1989, when an decrease in salinity of 3 °/oo was observed at the surface compared to deeper waters.

3.3.2 Temperature

Temporal variation in surface and bottom temperatures for the Graving, Albert and Queens Docks are illustrated in figs. 3.1A, 3.2A and 3.3A. Recorded temperatures in the South Docks ranged from 23.4 ^{o}C (26/7/89 - Queens) to 5.6 ^{o}C (2/3/90 - Queens) over the sampling period. Queens Dock showed the most extreme temperatures in all seasons, although the variation between docks was rarely greater than 2 ^{o}C. During the cold spell of early 1991 casual records of temperature showed that the water fell to 2.0 ^{o}C

When comparing the incidence of thermal stratification between years it should be borne in mind that sampling in 1988 did not begin until the end of May, one month after the onset of stratification in other years. For the purpose of quantifying the occurrence of thermal stratification the definition of a thermocline according to Birge (1887) is used, (i.e. greater than 1 ^{o}C temperature change over 1m depth). This definition is somewhat arbitrary but allows some comparison of frequency of occurrence, whereas later definitions (e.g., plane of maximum rate of decrease in temperature - Hutchinson 1957) do not.

In the Graving Dock marked thermal stratification was apparent for the month prior to experimental mixing. Thermal stratification was most marked on the 21st June 1988 when a 7.2 ^{o}C drop in temperature was recorded between 1m and 3 m depth. Artificial mixing began on the 1st July and thermal stratification was eliminated within 5 days. Thermoclines were recorded on only 2 occasions over the rest of the sampling period with the water mixer in operation, although a gradual decline in temperature from surface to bottom was a common occurrence in summer months. During periods of malfunction or experimental disuse of the mixer in spring and summer stratification soon reappeared.



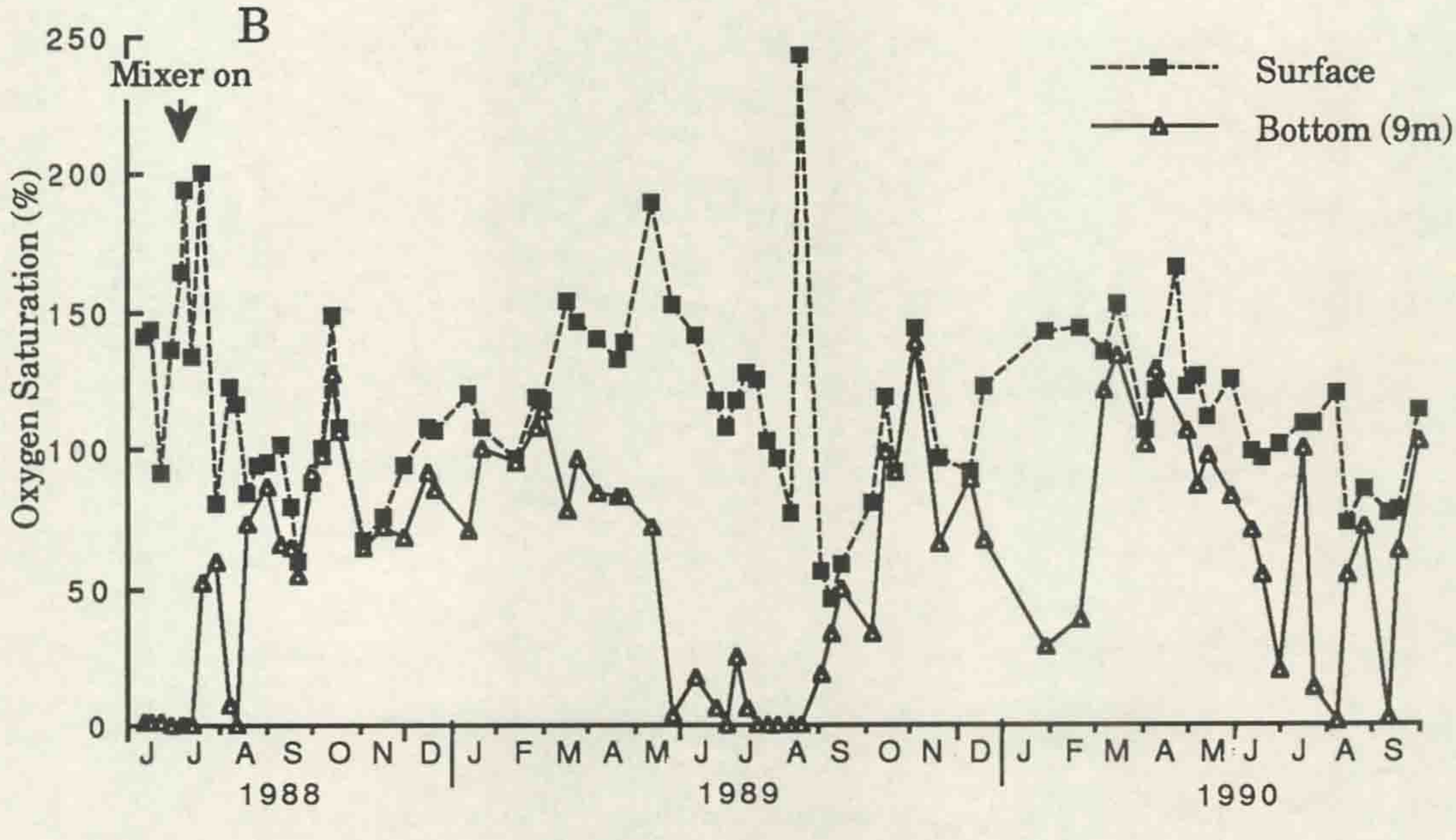


Fig. 3.1 Graving Dock temperature (A) and dissolved oxygen (B) at dock surface and bottom. May 1988 to September 1990. Horizontal bars indicate periods when mixer was switched off.

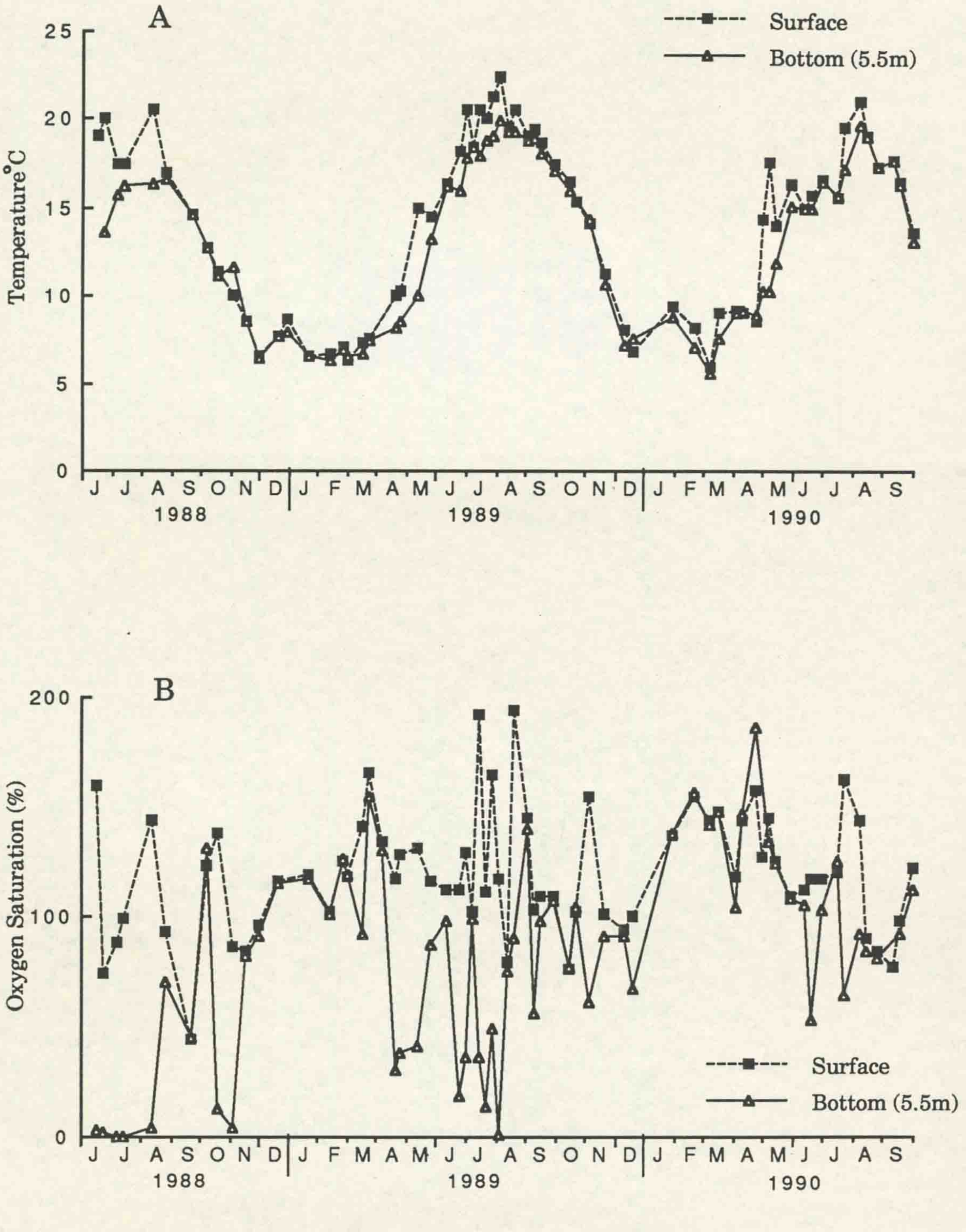


Fig. 3.2 Albert Dock temperature (A) and dissolved oxygen (B) at dock surface and bottom. May 1988 to 1990

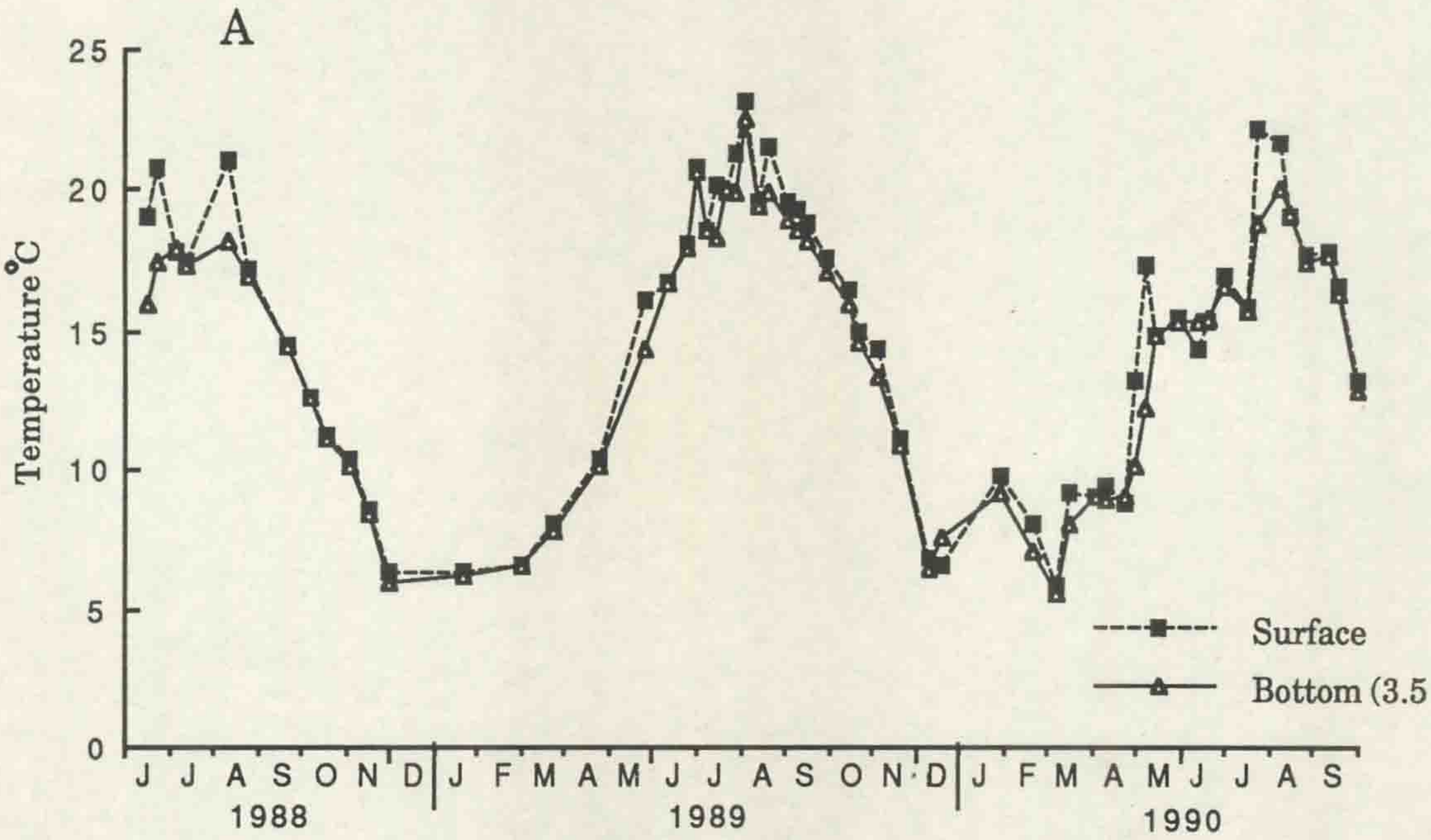


Fig. 3.3 Queens Dock temperature (A) and dissolved oxygen (B) at dock surface and bottom. May 1988 to September 1990.

The Albert Dock was stratified at the start of sampling (26/5/88) and this state persisted for 4 to 5 weeks. In 1989 thermoclines were recorded occasionally from the beginning of May until the end of July, persisting for a maximum of between 1 and 2 weeks. In 1990 the only period of thermal stratification recorded occurred in late April and persisted for just over 2 weeks.

Despite its shallow and open nature Queens Dock was not immune to thermal stratification. Thermoclines were recorded in June and August 1988, August 1989 and April and July 1990. The maximum recorded duration of thermal stratification was between 1 and 2 weeks.

3.3.3 Dissolved Oxygen

Temporal variation in surface and bottom water oxygen saturation is shown in figs. 3.1B, 3.2B and 3.3B for the Graving, Albert and Queens Docks respectively. Supersaturated surface waters (up to 245%) and anoxic (devoid of dissolved oxygen) bottom waters were recorded periodically in all docks.

In the Graving Dock, before installation of the mixer, dissolved oxygen was typically supersaturated in surface water, decreasing rapidly with depth to meter zero below 4m. Installation of the mixer improved the dissolved oxygen levels considerably, with more than 60% saturation in water above 6m occurring at all times with continual mixing. In the hot summer of 1989 low oxygen levels were frequently seen in bottom waters despite continuous mixing, but in summer 1990 periods of anoxic conditions were only recorded when the mixer was switched off for experimental reasons. During the winter months oxygen concentrations usually remained high at all depths, without the need for artificial aeration. The exception to this was seen in January and February 1990 when oxygen saturation above the sediments fell to less than 50%. Further details of the effects of the mixer are given in chapter 6.

The highest observed dissolved oxygen saturation was recorded in the Graving Dock in

August 1989 when the mixer was turned off during an experiment. An oxygen saturation of 245% was indicated which should be treated with caution as this is outside the accurate range of the oxygen electrode.

Dissolved oxygen levels in the Albert Dock improved over the three summers monitored (see fig. 3.2) with saturations of less than 20% being recorded for a maximum duration of more than 2 months in 1988 and two to three weeks in 1989. In 1990 these low levels were not observed.

In the Queens Dock depletion of oxygen to below 20% saturation in deeper waters were seen in both 1988 and 1989, lasting a maximum of one to two weeks. As in the Albert Dock, no such low values were seen in summer 1990.

3.3.4 Water Clarity

Secchi extinction depths varied between 8.5 m and 0.5 m with the highest values recorded in the Graving Dock and poorest clarities generally occurring in Queens Dock. In both the Albert and Queens docks maximum possible Secchi extinction depths were recorded each winter, the Secchi disc being visible resting on the dock bottom. Marked seasonal differences were seen with the water clarity being generally highest from December to February and lowest from June to August (fig. 3.4). In 1989 water clarity was very poor in March.

In summer 1988 water clarity was low at all sites. Greatly increased water clarity was apparent in the Albert and Graving Docks in 1989 and 1990. In the Queens dock no such improvements were seen. These results are covered in more detail in chapter 6.

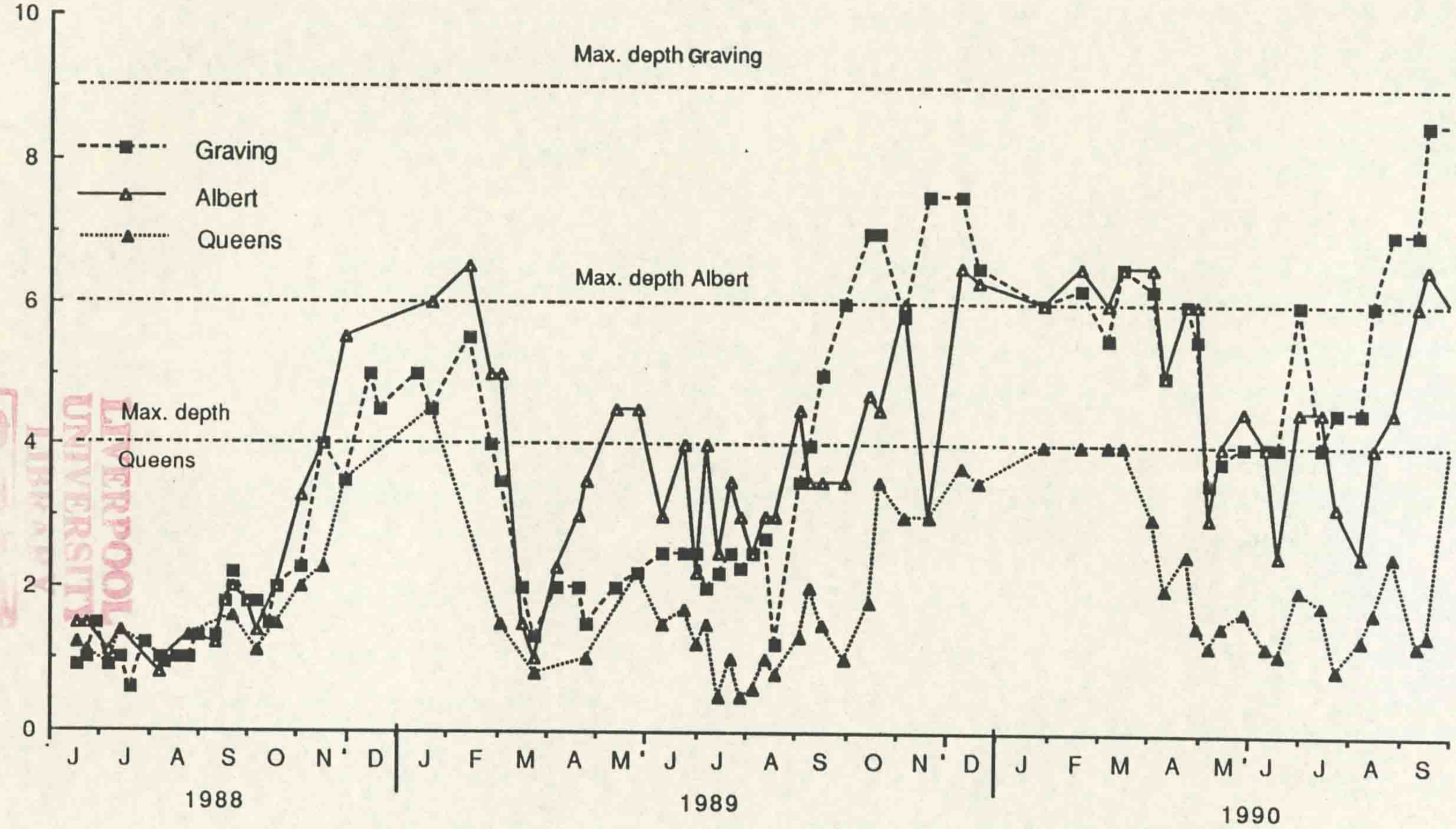


Fig. 3.4 Water clarity (Secchi extinction depth in m) Graving, Albert and Queens Docks May 1988 to September 1990. Dashed horizontal lines indicate normal max. depth of Graving (9m), Albert (6m) and Queens (4m) Docks.

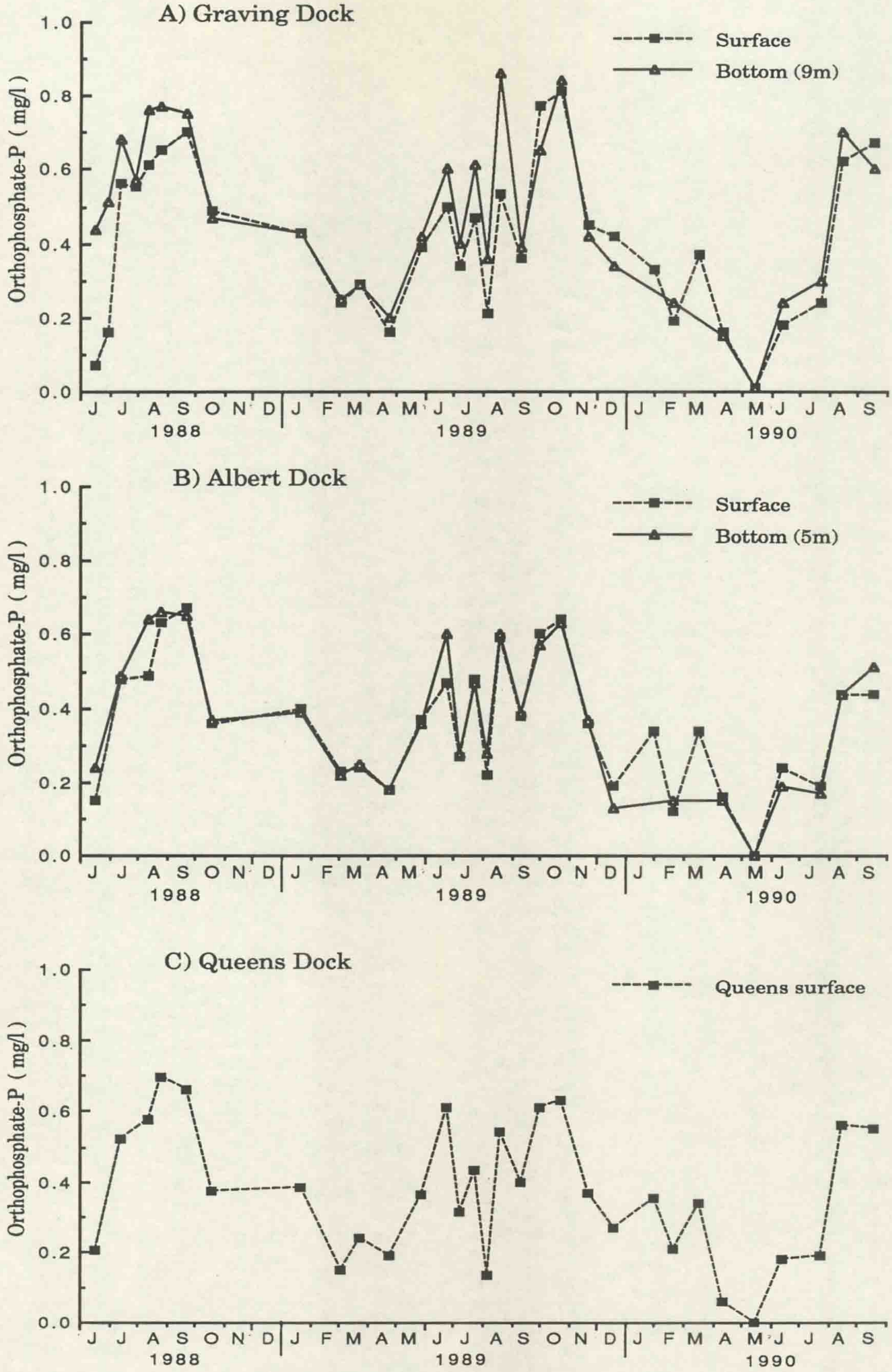


Fig. 3.5 Orthophosphate concentrations in Graving (A), Albert (B) and Queens Dock (C). June 1988 to September 1990.

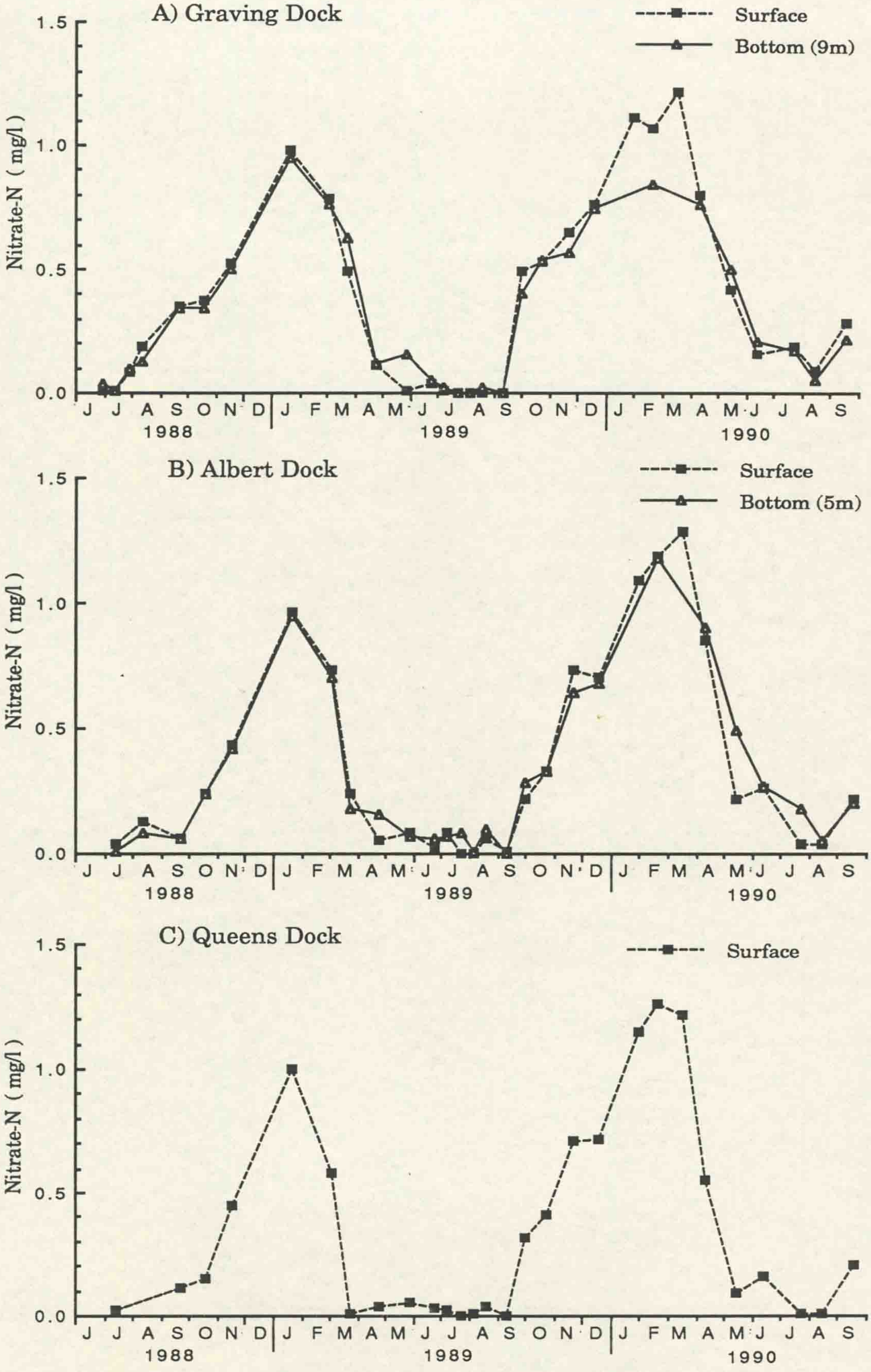


Fig 3.6 Nitrate concentrations in Graving (A), Albert (B) and Queens (C) Dock. June 1988 to September 1990.

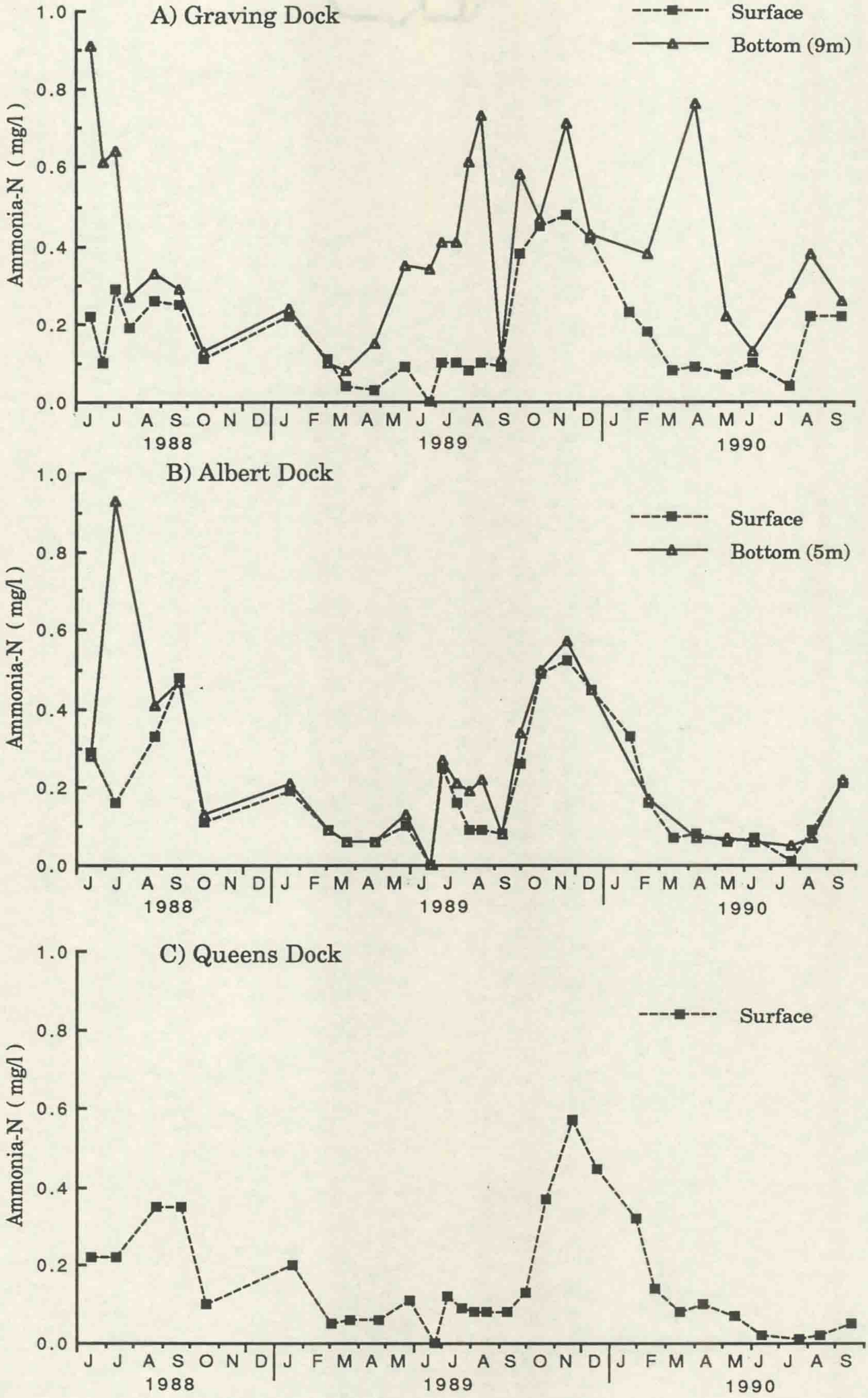


Fig. 3.7 Ammonia concentrations in Graving (A), Albert (B) and Queens Docks. June 1988 to September 1990.

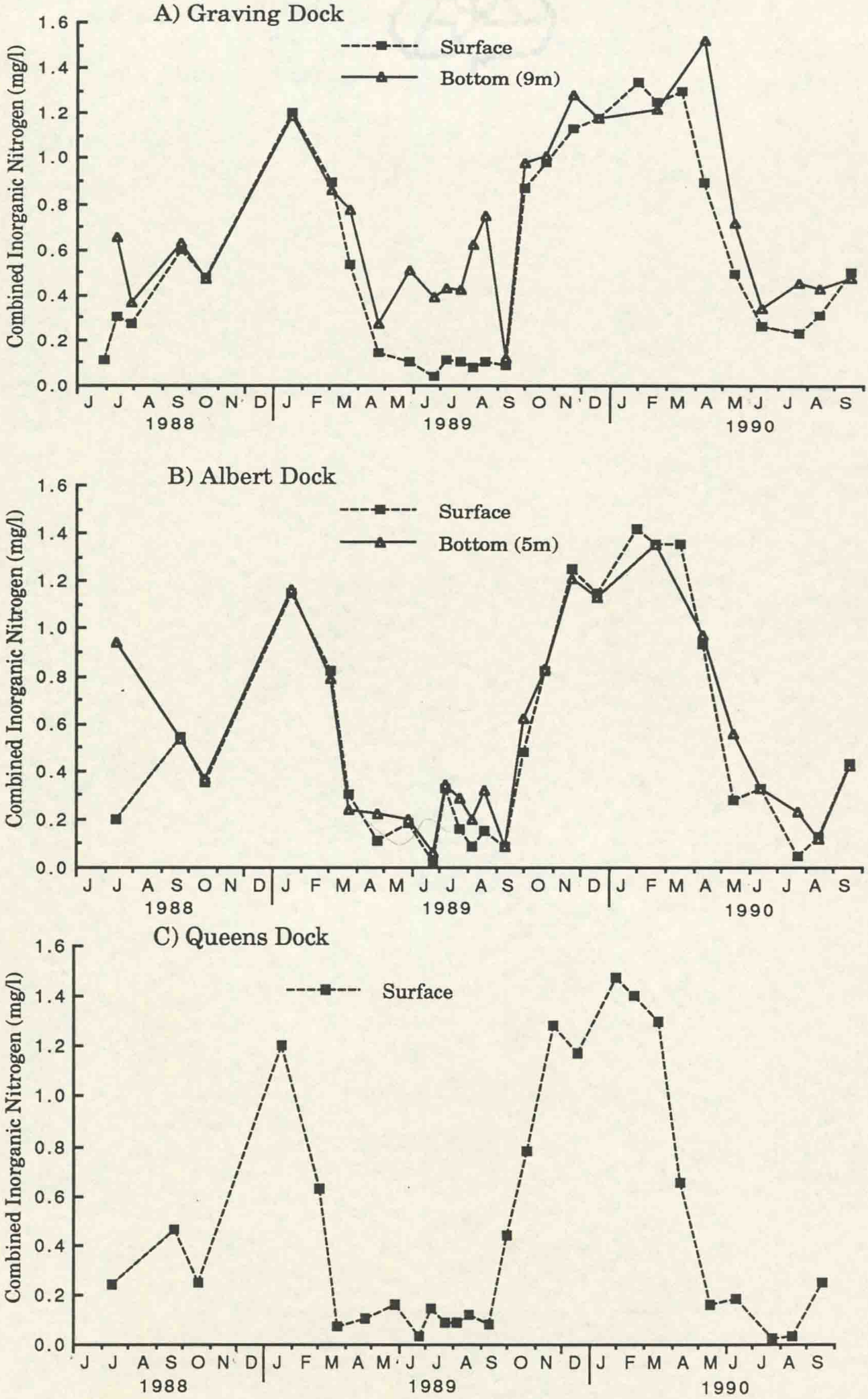


Fig. 3.8 Combined inorganic nitrogen concentrations Graving (A), Albert (B) and Queens (C) Docks. June 1988 to September 1990.

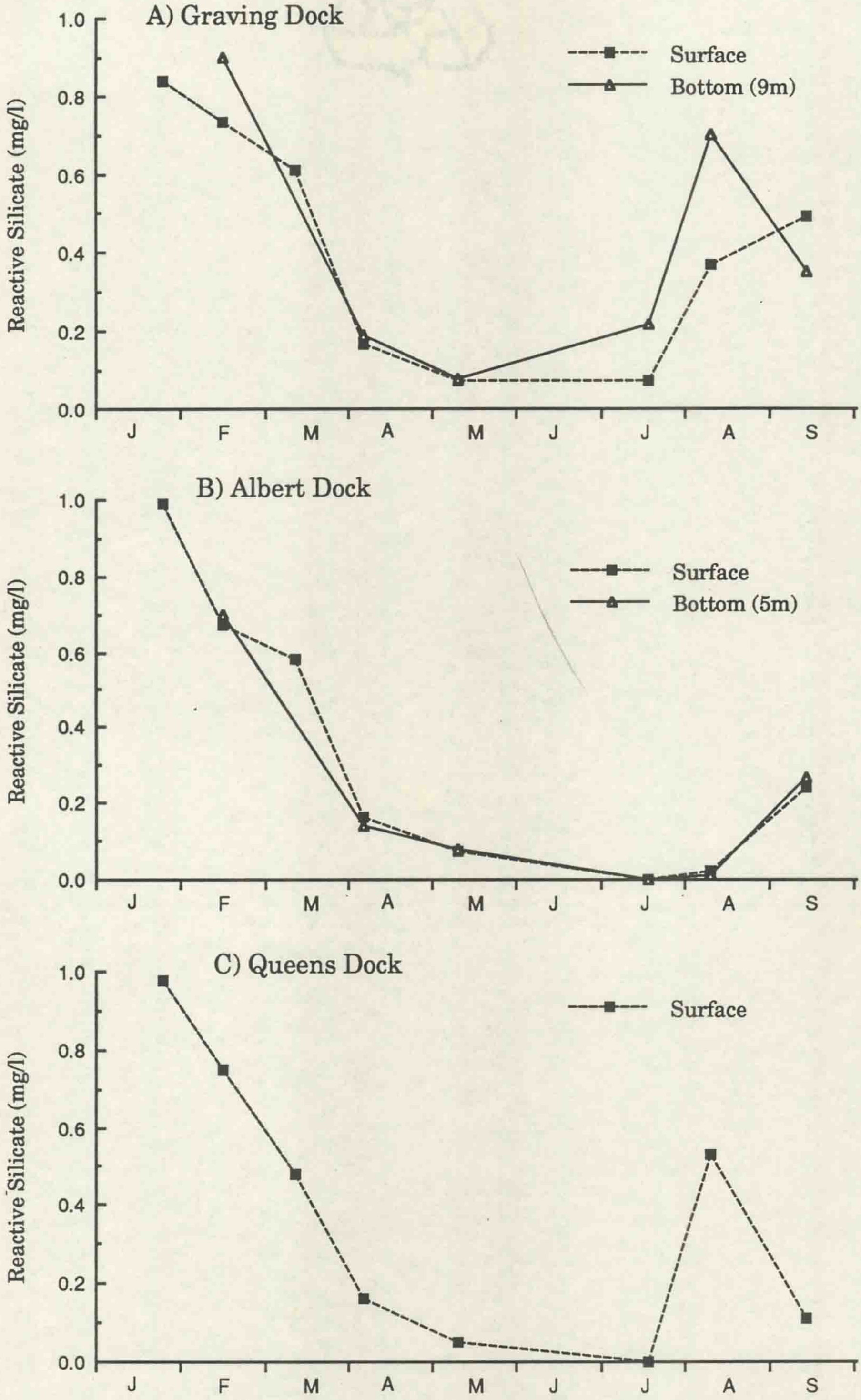


Fig. 3.9 Reactive silicate concentrations in Graving (A), Albert (B) and Queens Dock. January 1990 to September 1990.

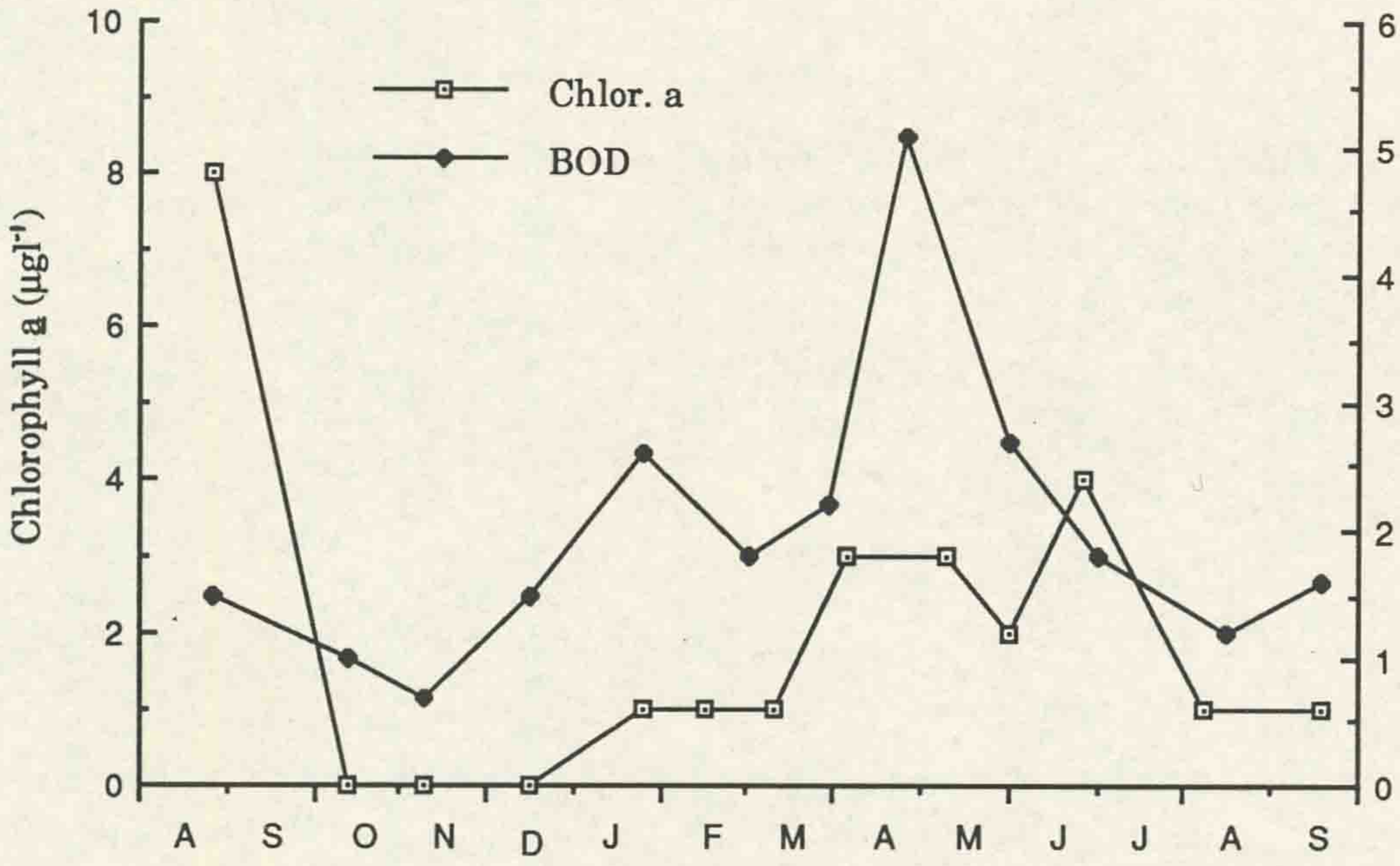


Fig. 3.10 Biochemical Oxygen Demand and chlorophyll *a* concentrations in the Albert Dock, August 1989 to September 1990.

3.3.4 Nutrients

Figs 3.5 to 3.9 show fluctuations in orthophosphate, nitrate, ammonia, total inorganic nitrogen and silicate in the three docks from June 1988 to October 1989. Variation between replicate samples was very small and often undetectable (see appendix Tables I to IV). Similar concentrations and changes with time were seen in all three docks, with spring/ summer reductions seen in all major plant nutrients. Maximum and minimum values for plant nutrients over the sampling period are summarised in table 3.1.

Table 3.1. Maximum and minimum recorded concentrations of plant nutrients in the Graving, Albert and Queens docks, June 1988 to September 1990. Bracketed values indicate missing value on date of maximum recorded concentration at other sites, due to malfunction of collecting bottle.

	Nitrate - N mgl^{-1}	Ammonium - N mgl^{-1}	Total Inorganic-N mgl^{-1}	Ortho-phosphate - P mgl^{-1}	Reactive Silicate - Si mgl^{-1}
Graving Dock					
Surface	<0.01 - 1.11	<0.01 - 0.48	0.04 - 1.25	<0.01 - 0.81	0.07 - 0.84
5m	<0.01 - 1.28	<0.01 - 0.78		<0.01 - 0.84	0.06 - 0.99
9m	<0.01 - 0.95	0.08 - 0.91	0.12 - (1.52)	<0.01 - 0.84	0.08 - (0.90)
Albert Dock					
Surface	<0.01 - 1.28	<0.01 - 0.52	0.02 - 1.42	<0.01 - 0.67	<0.01 - 0.99
5m	<0.01 - 1.18	<0.01 - 0.93	0.09 - (1.35)	<0.01 - 0.66	<0.01 - (0.70)
Queens Dock					
Surface	<0.01 - 1.26	<0.01 - 0.57	0.02 - 1.47	<0.01 - 0.66	<0.01 - 0.98

Phosphate concentrations were usually greater than 0.15 mgl^{-1} even during bloom conditions and only fell to below detection limits on one occasion (May 1990). The seasonal pattern was one of lowest concentrations in April/May, rising throughout the summer to peak concentrations in early autumn (Fig. 3.5).

Nitrate concentrations were low throughout spring and summer and reached a maximum in December/January (Fig. 3.6). Ammonia followed a more variable seasonal cycle, although highest concentrations were generally seen in the deeper docks in summer months (Fig. 3.7). Combined inorganic nitrogen showed a sustained reduction throughout spring and summer,

this was particularly marked from March to August 1989 (Fig. 3.8). Ammonia and nitrate concentrations examined individually fell below the detection limits of 0.01 mgl^{-1} on several sampling occasions. When all inorganic nitrogen fractions are combined however the lowest concentration recorded was 0.02 mgl^{-1}. Therefore the supply of nitrogen for phytoplankton growth was never completely exhausted.

Reactive silicate concentrations showed a steady spring reduction, remaining low until the end of summer (fig. 3.9). In the Queens Dock a drop in silicate level was also seen in September. The lowest silicate levels were seen in Queens and Albert Docks in July when concentrations were below the detection limit of 0.01 mgl^{-1}.

Variation in the concentration of some nutrients with depth was seen in the Graving and Albert Docks. No measurements of nutrients with depth were carried out in the Queens Dock due to its shallow nature. Ammonia showed the greatest variation in concentrations with depth. Greatly elevated concentrations of ammonia were seen in deeper waters during periods of anoxia. Phosphate concentrations were often higher in surface than bottom waters in summer, again often during periods of anoxia. This was most marked in the Graving Dock before the artificial mixer was used.

3.3.6 pH

Table 3.2 shows the ranges of pH recorded in the three docks sampled. The maximum values were all recorded from surface samples and were a result of supersaturated oxygen levels during algal blooms. The normal range of pH outside a bloom was 7.5 to 8.0.

Table 3.2- Range of pH over all water depths in the docks from June 1988 to December 1988, from January 1989 to December 1989 and from January to September 1990.

YEAR	Albert Dock	Graving Dock	Queens Dock
1988	7.4 - 9.0	7.2 - 9.3	7.4 - 9.0
1989	7.9 - 8.4	7.3 - 8.7	7.5 - 8.9
1990	7.3 - 8.3	6.4 - 8.4	7.6 - 8.4

3.3.7 Biochemical Oxygen Demand

BOD's during autumn and winter were generally very low (table 3.3), however, in spring and summer BOD's were much higher (Fig 3.10). BOD is positively correlated with peak periods of algal growth (Spearmans rank correlation r = 0.561, n = 12, p = <0.05). The highest BOD levels were recorded in Queens Dock.

Table 3.3 BOD (mgl^{-1} O_2 used in 5 days @ 20 °C) mean and range of monthly determinations 25/8/89 to 19/9/90.

SITE	MEAN	RANGE
Graving Surface water	2.1	0.2 - 4.4
Graving 5m	2.0	0.7 - 4.4
Graving 9m	2.0	0.5 - 3.6
Albert Surface water	2.0	0.7 - 5.1
Albert 5m	1.4	0.6 - 2.8
Queens Surface water	3.1	0.7 - 8.5

3.4 DISCUSSION

3.4.1 Ecosystem Functioning - The Physico-chemical Environment

The hydrography of the dock water is broadly governed by periodic intakes from its main water source, the Mersey Estuary. As described in chapter two this is only done at high spring tides, so intake water is of a fairly constant salinity and results in dock salinities varying from 23 - 28 °/oo , polyhaline conditions according to the Venice classification (Anonymous, 1959). All ground surface run-off is directed away from the docks. The exception is the Graving Dock which has stepped sides which will collect rainfall that can drain into the dock. The salinity stratification recorded in this dock in December 1989 was probably due to this run off.

The shallow nature of the docks allows rapid heating or cooling of the water body and, as may

be expected, temperature extremes are large. In 1989, however, the water temperature range just outside the docks, in the estuary, was greater than inside the docks (National Rivers Authority data), so organisms which survive in the estuary should be able to tolerate the dock temperature regime. The winters of 1988 and 1989 were very mild and lower water temperatures may occur in future years. Temperatures down to 0.3 ^{o}C were recorded in Sandon Dock, after which mortality of juvenile *Mytilus* was observed (Russell *et al* 1983). Severe winters may cause mortality in a wide range of species (Crisp 1964). Shallow water bodies such as the South Docks offer little protection from such temperature extremes.

The high summer water temperature extremes are likely to restrict the survival or growth of some marine species. A high loss rate and much reduced growth rates were observed in *Laminaria* plants introduced to Sandon Dock at temperatures greater than 15 ^{o}C (Russell *et al* 1983). The development of a *Laminaria* bed and associated community is therefore extremely unlikely in the South Docks.

All plant nutrients measured follow a seasonal cycle with spring or summer reduction in combined inorganic nitrogen, orthophosphate and silicate as nutrients are removed during the growth of phytoplankton. However annual cycles for individual nutrients are aphasic and minimum concentrations do not coincide with maximum phytoplankton biomass in every case. Concentrations of nutrients in surface waters follow a similar pattern in each dock.

Orthophosphate concentrations were at their lowest in spring, but rather than remaining low throughout the period of high phytoplankton biomass (March - August), as is the case with total inorganic nitrogen, a gradual rise in concentration was seen over the summer to a peak in early autumn, gradually decreasing over winter. This phenomenon was also observed in Lake Grevlingen (De Vries & Hopstaken 1984, Pellikaan & Neinhuis 1988). The most likely reason for this cycle is effects of phosphate flux from sediment to water with varying oxygen concentrations. The rate of phosphate release from the sediment is greater during periods

of low dissolved oxygen concentrations (Mortimer 1971) such as those seen in summer. The shallow nature of the docks enable sediment effects to have a large impact on the water column. In the Irish Sea where water volume per area of sediment and oxygen concentrations are greater, such effects are not seen, the annual cycle being lower levels in spring-summer and higher values from October to March, usually peaking in January / February (Slinn 1974, Slinn & Eastham 1984).

Total inorganic nitrogen concentrations appear to be closely related to the phytoplankton populations with peak concentrations at times of lowest biomass. Examining the individual components reveals that ammonia concentrations reached winter peaks earlier than nitrate and then tailed off. Ammonia is produced by mineralisation of decaying phytoplanktonic organic nitrogen, this ammonia will then be oxidised to nitrate by nitrifying bacteria (Webb 1981), hence the observed patterns.

In stratified conditions phytoplankton production becomes concentrated in the upper layers (Raymont 1980 for review) and consequently nutrients may be depleted in the epilimnion. Evidence of this was seen during periods of stratification in the Albert and Graving Docks when phosphate and total inorganic nitrogen became much lower at the surface than in deeper waters. In the case of phosphate this may also be enhanced by the increased release from the sediments in the anoxic conditions (Mortimer 1971, Patrick & Khalid 1974, Webb & D'Elia 1980). High concentrations of ammonia are also associated with anaerobic bottom waters during stratification events. Anaerobic mineralisation of organic nitrogen to ammonium continues under these conditions, but aerobic nitrification to nitrate cannot proceed and ammonia builds up in the hypolimnion (De Vries & Hopstaken 1984).

Low nutrient concentrations can limit phytoplankton growth following Liebig's law, in that yield is determined by the amount of the nutrient that is in minimal supply (Droop 1973). Nitrogen is commonly reported to be the most limiting nutrient in coastal and marine environments (Dugdale & Goering 1967, Ryther & Dunstan 1971, Slinn 1974, Carpenter &

Capone 1983, De Vries & Hopstaken 1984), whereas phosphate is normally more critical in freshwaters (Vollenweider 1968, Schindler 1981, Hecky & Kilham 1988). The supply of inorganic nitrogen fell to low levels throughout the summer months but was never completely exhausted in any dock. Silicate also reached very low levels in spring and summer, falling below the detection limits in some samples. Orthophosphate was below detection limits on one occasion but this may be an anomaly and should be treated with caution due to interference by salt at low levels. Low concentrations of a particular nutrient do not necessarily indicate nutrient limitation (Droop 1973) as supply by regeneration may be adequate to satisfy demands. Hence routine measurements of nutrient concentrations as carried out in this study cannot reliably determine whether a nutrient is limiting or not - experimental procedures such as nutrient additions or physiological analyses are required.

Paasche & Erga (1988) used both monitoring and experimental techniques in a study of nutrient limitation on natural phytoplankton assemblages in the polyhaline inner Oslofjord. They assumed that limitations could occur if concentrations fell below the rather arbitrary values of 1.0 μM (14.01 μgl^{-1}) for combined nitrogen salts and 0.1 μM (30.97 μgl^{-1}) for orthophosphate and found this to be generally consistent with experimental indicators. In the South Docks inorganic nitrogen concentrations reached or approached 1.0 μM on several occasions but minimal annual orthophosphate concentrations were normally around 6.0 μM hence, using the guidelines of Paasche & Erga, nitrogen occasionally limits phytoplankton growth whereas orthophosphate (barring one possibly anomalous result) does not.

Nitrogen requirements of phytoplankton, although variable, are roughly ten times that for phosphate by atoms (Ryther & Dunstan 1971). N rather than P limitation has been reported for ratios of 30:1 or less in laboratory experiments (Rhee 1978). Ratios of maximum inorganic nitrogen : silicate : orthophosphate are 0.8 : 0.8 : 1 by atoms in the South Docks, therefore nitrogen is in shorter supply in terms of requirements than phosphate. Silicate is a major nutrient only for diatoms (Webb 1981) being required by them in roughly the same proportions as nitrogen (Richards 1958, Redfield *et al* 1963, Stephens 1970). Therefore, in

the case of diatoms both nitrogen and silicate may limit growth. Silicate is also regenerated more slowly from decaying diatoms than nitrogen or phosphate, further restricting the supply for diatom growth (Ryther & Officer 1981). The argument that nitrogen and silicate are the more limiting nutrients is substantiated by the marked depletion in the concentrations of these nutrients throughout the period of phytoplankton growth which is not the case with orthophosphate.

3.4.2 Implications For Water Quality

Thermal stratification is a common summer phenomenon in poorly mixed water bodies such as the South Docks. Rapid heating of surface waters creates a less dense surface layer, increasing the stability of the water body and reducing the effect of wind mixing, so that a poorly oxygenated hypolimnion may result. This problem has been widely reported in various water bodies, such as fresh water lakes (Knoppert *et al* 1970, Pastorok *et al* 1981, Bailey-Watts *et al* 1987), fjords or shallow seas (Jørgensen 1980, Riisgard & Poulsen 1981) and in other docks (Russell *et al* 1983, Hendry *et al* 1988a, Conlan 1989).

Thermal stratification resulted in periods of low oxygen concentrations in all sampled docks. In the shallow Queens Dock the maximum period of low oxygen saturation (< 20%) was 1-2 weeks, in Albert Dock > 2 months and in Graving Dock > 3 months. Anoxia could therefore have a major impact on the benthic fauna in all docks. During periods of anoxia in a Danish fjord, tube polychaetes survived for up to 1 week while *Mytilus* and other lamellibranchs survived for 1-2 weeks (Jørgensen 1980). In laboratory experiments Theede *et al* (1969) found 50% mortality in hypoxic (0.15 ml O_2 l^{-1}) water at 10 °C after 2 hours for *Crangon crangon*, 2 days for *Carcinus maenas* , 5 days for *Nereis diversicolor* and 35 days for *Mytilus edulis*.. All are typical dock species. Mass mortality of sediment dwelling fauna would therefore be expected during the worst periods of anoxia in all docks, although *Mytilus* would be likely to survive in Queens Dock. Observations made by SCUBA diving during anoxic conditions showed dead and dying polychaetes on the mud surface and flatfish in respiratory distress

congregating in the shallower areas of Albert Dock. Occasional dead fish floating on the water surface have been reported during problem periods. Such sights are damaging to public perception of the water quality.

Low oxygen concentrations in bottom waters stimulate the anaerobic bacterial reduction of sulphate and putrefaction of proteins resulting in hydrogen sulphide build up (Theede 1969). During long periods of anoxia the strong 'bad eggs' smell of this gas was occasionally released when the water was disturbed. This is obviously undesirable in a development such as the Albert Dock. Hydrogen sulphide is toxic to marine fauna, particularly when associated with low oxygen concentrations and has been found to reduce survival times in anoxic conditions by 30 to 40% at concentrations of a few mgl^{-1} (Jørgensen 1980).

Supersaturated oxygen concentrations are frequently recorded in the South Docks both in surface layers in stratified conditions and throughout the water body in well mixed situations. Supersaturated conditions are produced by oxygen release from phytoplankton during periods of high primary production (Raymont 1980), and are frequently reported in eutrophic waters (e.g. Winter *et al* 1975, Umamaheswara Rao & Monanchand 1988). Oxygen depletion in such waters during hours of darkness due to respiration of phytoplankton may occur (Olson 1932). In a 24 hr study carried out in the Graving Dock in stratified bloom conditions this did not occur in surface waters, oxygen concentrations ranging from 13.2 - 12.2 mgl^{-1}. At 2.5 m depth, however, the variation in oxygen concentration was 9.5 - 0.18 mgl^{-1} illustrating a rise in the oxycline. In shallow waters such as the Queens Dock the night-time oxygen sag may cause benthic organisms to be subjected to anoxic conditions, although oxygen concentrations, when monitored in the day-time, would appear to be good.

Thermal stratification in the Albert Dock was weaker in the summers of 1989 and 1990 compared to 1988. Weaker stratification certainly accounts for a proportion of the dramatic improvements in hypolimnetic oxygen concentrations over this period. However, in 1988 low oxygen levels were noted without the presence of thermal stratification and in May 1990

oxygen depletion did not occur despite a 2-3 week period of thermal stratification. This reduced tendency for the development of anoxic conditions may be a result of the absence of dense phytoplankton blooms during the summers of 1989 or 1990. Algal blooms can cause dissolved oxygen depletion during decomposition and decay (Reynolds and Walsby 1975). The reduction in phytoplankton biomass may be due to the filtering action of the dense *Mytilus* population which colonised the Albert Dock in September 1988. It is also possible that thermal stratification itself was reduced as a consequence of decreased phytoplankton biomass. In the oceans the absorption of solar radiation is dependent to a large extent on the concentration of photosynthetic pigments (Smith & Baker 1978). It has been shown that increased absorption of radiation due to increased phytoplankton biomass enhances the rate of heating at the ocean surface and reduces distribution of heat through the water column (Sathendranath *et al* 1991). Hence the decrease in phytoplankton biomass seen in 1989 and 1990 compared to 1988 (see chapter 4) may have resulted in more uniform heating of the water body and less thermal stratification.

In the Graving Dock long periods of low hypolimnetic oxygen were not eliminated by continuous artificial aeration, although thermal stratification was generally prevented. The mixer was underpowered and oxygen demands of water and sediments outstripped its ability to aerate the water body.

Turbidity of the water in the South Docks is determined almost entirely by phytoplankton populations rather than other suspended material, water clarity being negatively correlated with chlorophyll *a* (Spearmans Rank Correlation $r = -0.711$, $n = 26$, $p = <0.001$). Suspended inorganic loads in the dock waters are very low due to the lack of strong water currents or land run-off. Water clarity may be very low in summer causing the water quality to look very poor.

Water clarity was significantly improved in 1989 and 1990 compared to 1988 in the Albert and Graving Docks whereas Queens Dock showed little improvement (see chapter 6, Table

6.4). Increases in water clarity are thought to be linked to increased filter feeding pressure. This is covered in detail in chapter 6.

The main water quality problems in the South Docks (nuisance phytoplankton blooms, hypolimnetic oxygen depletion and poor water clarity) are typical symptoms of a eutrophic water body (Jaworski & Villa 1981). Vollenweider (1968) defines eutrophication as 'the enrichment of a water body in nutrients and the ensuing deterioration of their quality due to the luxuriant growth of plants with its repercussions on the overall metabolism of the waters'. Therefore, in understanding the water quality problems at the South Docks and assessing the potential for future improvements, consideration of nutrient status and possible long term changes is important. As previously discussed, nitrogen and silicate are the most limiting nutrients so any management strategy should pay particular attention to reduction of these nutrients in order to control phytoplankton growth.

Anaerobic waters may result in an increase in the total inorganic nitrogen and orthophosphate present in the water column. In reducing conditions nitrification and hence the supply of nitrate for denitrification is inhibited, consequently the conversion of nitrate to gaseous products and hence loss to the atmosphere will cease (Webb 1981). Denitrification has been found to remove on average 60% of the total N loading in lakes (Vollenweider 1968) and 20-50% in estuaries (Seitzinger 1988). Anaerobic water may lead to a marked increase in total nitrogen in the system (De Vreis & Hopstaken 1984). Phosphate associated with manganese and iron in the sediments is also released in reducing conditions and can be the major source of dissolved phosphate at certain times of the year (Mortimer 1971, Hallberg *et al* 1976, Marsden 1989).

The low autumn/winter Biochemical Oxygen Demand with higher spring and summer values indicates that the breakdown of organic material from phytoplankton die off contributes more to oxygen depletion than other sources of organic material, such as detritus brought in with top up water, which is available all year round. BOD was higher in Queens Dock than

in the Albert and Graving Docks, probably due to the higher phytoplankton biomass present in Queens Dock.

In summary, the South Docks are a brackish water (polyhaline), nutrient rich water body which is prone to extremes of temperature due to its shallow nature. Thermal stratification was commonly recorded in summer months, particularly in the deeper docks. The main physical or chemical water quality problems are low dissolved oxygen levels in summer, which are normally associated with thermal stratification, and high concentrations of plant nutrients.

CHAPTER FOUR

TEMPORAL AND SPATIAL VARIATIONS IN PLANKTON

4.1 INTRODUCTION

The dynamics and composition of plankton communities are an important consideration in understanding the general ecology and water quality characteristics of an aquatic ecosystem. Phytoplankton, whilst being the major primary producers in many aquatic food chains, may grow to nuisance levels in nutrient rich waters such as the South Docks.

Phytoplankton-related water quality problems are a common occurrence in other British Docks and in hydrographically similar water bodies throughout the world. Nuisance phytoplankton blooms are a common problem in enclosed/semi-enclosed fresh and oligohaline waters (e.g. Ryther & Dunstan 1971, Reynolds & Walsby 1975, Crawford 1979) and in saline lagoons and coastal waters (e.g. Mahoney & Steimle 1979, Cross & Southgate 1980, Silva 1985, Bätje & Michaelis 1986). Nuisance species are normally cyanobacteria in fresh and oligohaline waters, whilst in marine systems dinoflagellates cause most concern (Paerl 1988). In several restored docks phytoplankton blooms have led to discoloration of the water, high turbidity or algal scums, all detracting from the high water quality standards required by the redevelopment schemes (Hendry *et al* 1988, Conlan 1989, Conlan *et al* in press, Hawkins *et al* in press b). The toxicity of some bloom species has more serious implications for dockland developments. Blooms of toxic algae have caused mortality of aquatic fauna (Boalch 1979, Cross & Southgate 1980, Jones *et al* 1982) and have resulted in respiratory and skin irritation amongst watersports users (Tufts 1979, Carmichael 1981, Pease 1989). Mortality of aquatic fauna may also be associated with blooms of non-toxic phytoplankton due to oxygen depletion during algal die-off (Mahoney & Steimle 1979).

Relatively little information is available on the plankton of disused docks, particularly those of higher salinity. Hendry *et al* (1988) gives a brief account of the main phytoplankton species and zooplankton groups found in ten disused docks throughout the U.K. Conlan (1989) contains a more detailed, quantitative study of phytoplankton and zooplankton in an oligohaline, lower estuarine dock at Preston, Lancashire. Descriptions of the plankton of

enclosed waters of similar salinity to the South Docks have been carried out for phytoplankton (e.g. Pratt 1965, Bakker & De Vries 1984, Rijstenbil 1987) and zooplankton (e.g. Pratt 1965, Collins & Williams 1981, Taylor 1987).

Seasonal variation in phytoplankton biomass is primarily dependent on light availability (Fogg 1975, Barnes & Mann 1980) and to some extent water temperature (Fisher *et al* 1988). Species composition and biomass of phytoplankton communities may also be influenced by other physico-chemical factors, for example, nutrient concentrations (Eppley *et al* 1969, Webb 1985), salinity (Rijstenbil 1987), mixing (Bakker & De Pauw 1974, Rijstenbil 1987, Paerl 1988) and toxic contaminants (Sanders et al 1981, Ryther & Officer 1981). Biological factors such as zooplankton grazing (Pratt 1965, Ryther & Officer 1981) and benthic filter feeders (Officer *et al* 1982, Hily 1991).

The growth and reproduction rate of zooplankton is normally temperature dependant (Orsi & Mecum 1986, Patalis & Salki 1984) and may also be closely correlated with phytoplankton populations if this is their major food source (Canfield 1984). As phytoplankton are primarily seasonally light-limited there is a lag phase at the beginning of spring, when light levels but not temperature have increased, and phytoplankton populations can grow relatively unchecked by grazing pressure (Barnes & Mann 1980).

Phytoplankton and zooplankton communities were studied over a 28 month period in the Graving, Albert and Queens Docks. The specific aims of the study were, firstly to identify annual and longer term trends in plankton biomass and species composition in the various docks. Secondly, in line with the required emphasis on water quality, particular attention was given to the dynamics of phytoplankton blooms and identification of potentially toxic species. The results obtained in this chapter were then related to the hydrography described in the previous chapter.

4.2 METHODS

4.2.1 Chlorophyll *a*

Chlorophyll *a* was measured as an indicator of phytoplankton biomass. Water samples for chlorophyll *a* determination were collected concurrently with samples for nutrient analysis as described in section 3.2.2. Three replicate 0.5 l samples were taken for each site/depth. These were filtered immediately on returning to the laboratory through 7 cm GF/C filter papers onto which a few drops of magnesium carbonate suspension had been added to reduce acidity on the filter paper. In winter months, when chlorophyll *a* concentrations were low, it was necessary to pool the three replicate samples in order to provide sufficient material for detection. Filter papers were then frozen and analysis carried out within 30 days. Chlorophyll *a* concentrations were determined using the slow acetone method with correction for phaeophytin (HMSO 1980).

4.2.2 Phytoplankton

Water samples for phytoplankton identification and enumeration were collected in the same manner as those for nutrient samples described in section 3.2.2. Three replicate 150 ml samples per site depth were placed in poly-ethylene bottles and preserved with Lugols iodine immediately on returning to the laboratory. Preserved samples were settled out in 5 ml or 25 ml chambers, depending on cell concentration and counted using an inverted microscope (see Lund *et al* 1958, Hasle 1978). Enumeration was carried out at x400 magnification from fields of known area selected randomly from parallel transects across the chamber floor, ensuring even counting over the chamber floor to reduce errors due to non-random settling of cells. Brief examination of the chamber at x100 magnification was carried out and the presence/absence of any very large cells, missed by previous counting, was noted.

Initially it was intended to count three subsamples from each of three replicate samples, but this proved to be too time consuming and it was necessary to reduce the number of counts. A pilot study was carried out to determine the greatest source of variation in counting in order to plan the most effective concentration of effort. A nested ANOVAR of between replicate and

between sub-sample variance found that only 9% of the total variance to be due to sub-sampling. It was therefore decided to count only one settled chamber (sub-sample) per replicate sample and to collect three replicates from each site / depth. A minimum of 100 algal cells (excluding 4 - 10 μm category) were counted from a minimum of 50 fields of view, this was considered sufficient to give a reasonable assessment of diversity and cell numbers (Lund *et al* 1958). As the area of the fields of view and total area of the chamber floor were known the concentration of algal cells could then be calculated. Identification of species > 10 μm diameter was carried out to at least genus level. Major bloom organisms were identified to species, with examination of live material under phase contrast optics or using S.E.M. techniques when necessary. Phytoplankton cells 4 to 10 μm diameter were grouped simply as 'small monads and flagellates' and those <4 μm diameter were not enumerated. In a study of 15 sets of replicate (n=3) samples enumerated in this way the maximum / minimum values were always within 30% of the mean, an acceptable reliability where changes in cell concentrations of several orders of magnitude are observed.

Biomass of the major phytoplankton species (15 species) was estimated from measurements of appropriate length, width and height components. A geometric approximation was then made for each species and cell volume calculated using appropriate formulae. Measurements of each species were carried out on at least two occasions and an average taken. Errors will be introduced due to the variability of cell volume of species throughout the year but this is small compared to annual variations in biomass. The average density of the phytoplankton was assumed to be 1 g ml^{-1} (Reynolds 1986) and biomass as mg wet weight l^{-1} was calculated.

Surface samples were collected from Graving, Albert and Queens Docks at monthly intervals from June 1988 to September 1990 with additional samples taken at times of particular interest. Samples from 5m depth were taken from the Albert Dock at monthly intervals from July 1988 to September 1989. Changes in phytoplankton populations with depth under different mixing regimes were studied in the Graving Dock and are described in chapter 6.

A detailed study of the spring to summer phytoplankton succession was carried out in the Queens Dock from 19th March to 14th May 1990. Samples were taken from surface waters once at mid-morning every two to three days whenever possible.

4.2.3 Zooplankton

Zooplankton was sampled initially by means of a 140 µm mesh net but this was found to clog quickly so a 250 µm mesh net of 30cm diameter was chosen for continued sampling. Three replicate hauls were taken from bottom to surface in each of the three docks. Samples were preserved on site in 4 % formalin-seawater buffered with borax. Initially replicate hauls were enumerated separately to allow assessment of reliability of results. However, due to a dramatic decline in zooplankton numbers it became necessary to pool the three hauls. In a study of 11 sets of replicate (n=3) samples the maximum and minimum counts were found to be within 30% of the mean within replicate sets, compared to temporal variation between samples which spanned six orders of magnitude.

Samples were stained with rose bengal and examined at x 25 magnification in a grooved counting tray. Identification was carried out to species level wherever possible. The large dinoflagellate *Noctiluca scintillans* was enumerated along with zooplankton samples.

4.3 RESULTS

4.3.1 Chlorophyll *a*

The variation in mean chlorophyll *a* concentrations in Graving, Albert and Queens Docks over the sampling period is shown in fig. 4.1. Agreement between replicate samples was usually good, for reasons of clarity error terms are omitted from these graphs but are given along with means in appendix Table V.

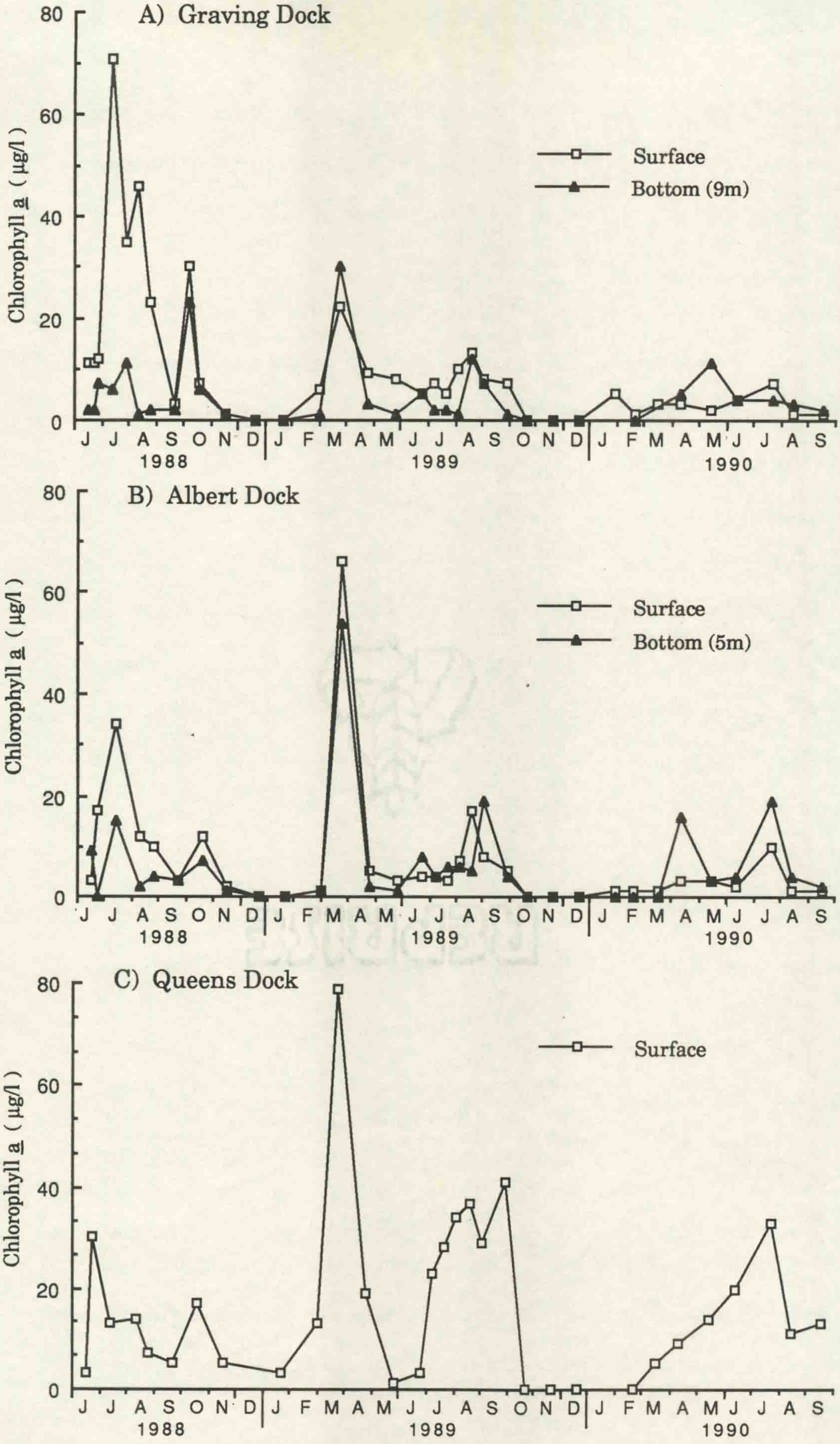


Fig. 4.1 Chlorophyll a Concentrations, Graving (A), Albert (B) and Queens (C) Docks. June 1988 to September 1990.

Maximum chlorophyll *a* concentrations in the Albert (66 μgl^{-1}) and Queens (79 µgl-1) Docks were recorded in March 1989 during a diatom / *Phaeocystis pouchettii* bloom. Maximum chlorophyll *a* concentrations in the Graving Dock (71 μgl^{-1}), occurred just after the start of artificial mixing during a bloom of the euglenoid *Eutreptiella*. The large chlorophyll *a* spring peak seen in 1989 did not recur in 1990, chlorophyll concentrations rising steadily throughout the spring in all docks. Winter chlorophyll *a* concentrations were frequently below detection limits in all docks in winter months (November - February)

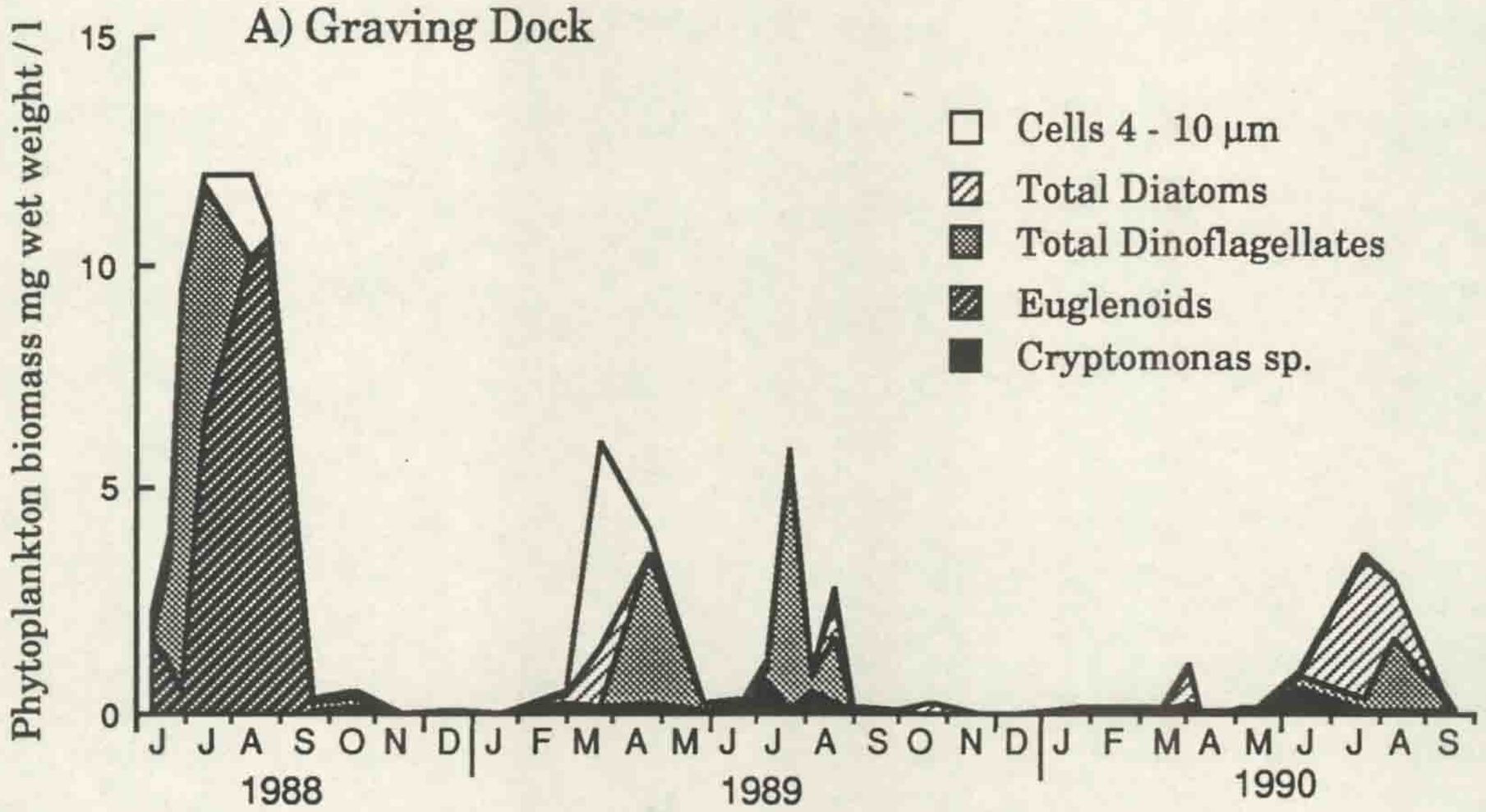
In the surface waters of the Albert and Graving Docks the summer (June - September) chlorophyll maxima was higher in 1988 than 1989 or 1990, this was particularly marked in the Graving Dock. In Queens Dock a similar summer chlorophyll maxima occurred in all years

Chlorophyll *a* concentrations were often much higher in surface samples than bottom samples in the Graving and Albert Docks in summer 1988. This situation was most notable in the Graving Dock where it persisted even after the onset of artificial mixing. Strong vertical gradients of this kind were not observed in summer 1989 and 1990. Interestingly, in the Albert Dock in summer 1990, higher concentrations were more frequently observed in bottom samples.

4.3.2 Phytoplankton

4.3.2.1 Spatial, Annual and Long Term Variation

Variation in phytoplankton biomass, as calculated from cell volume, in the surface waters of the Graving, Albert and Queens Docks surface waters is illustrated in Fig. 4.2 A to C. Phytoplankton species lists and cell concentrations for all samples are given in the appendix (Tables VI toVIII). Variations in total phytoplankton concentrations over the sampling period, for surface waters in the Graving, Albert and Queens Docks, are illustrated in Fig 4.3.



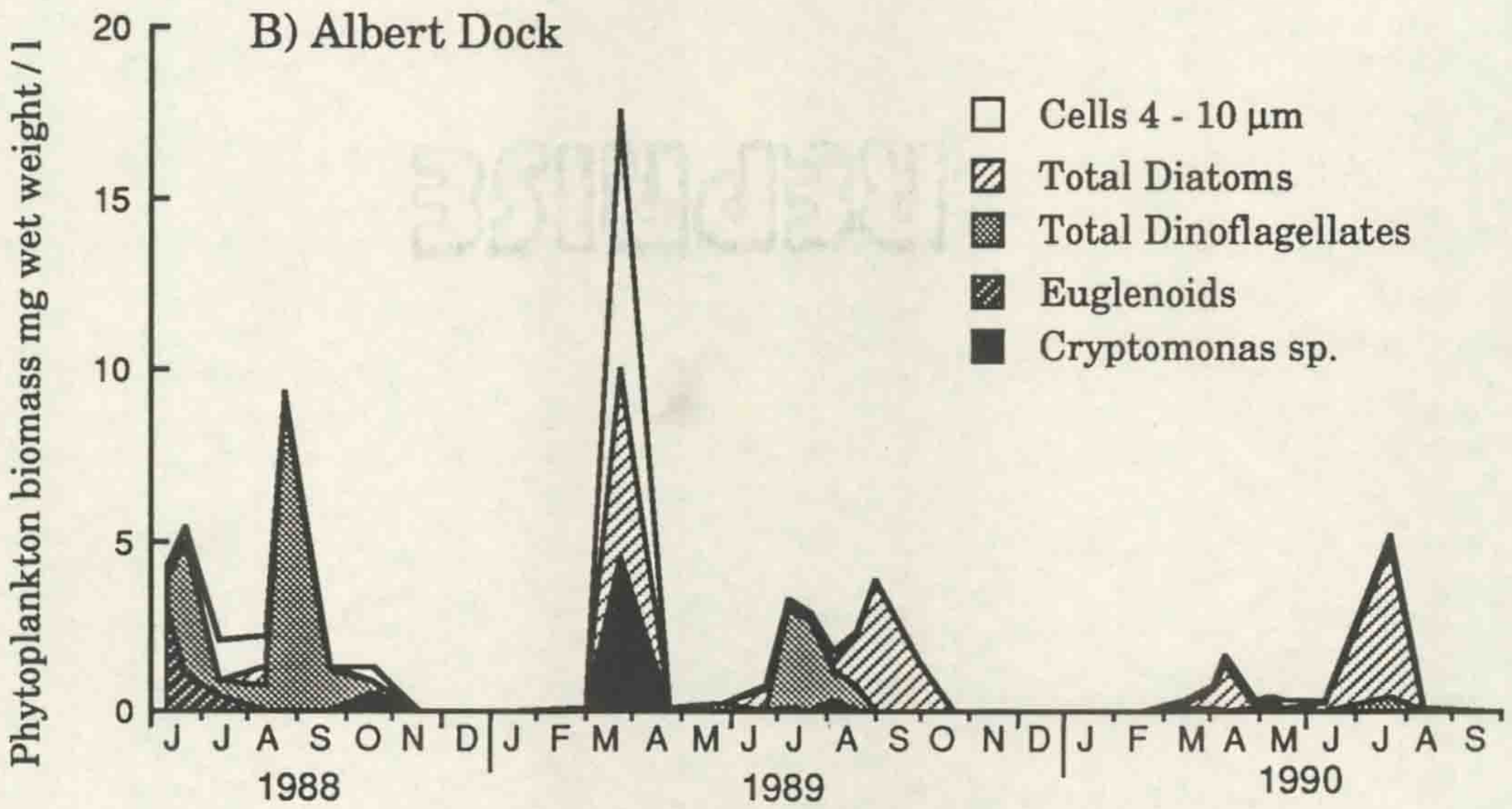
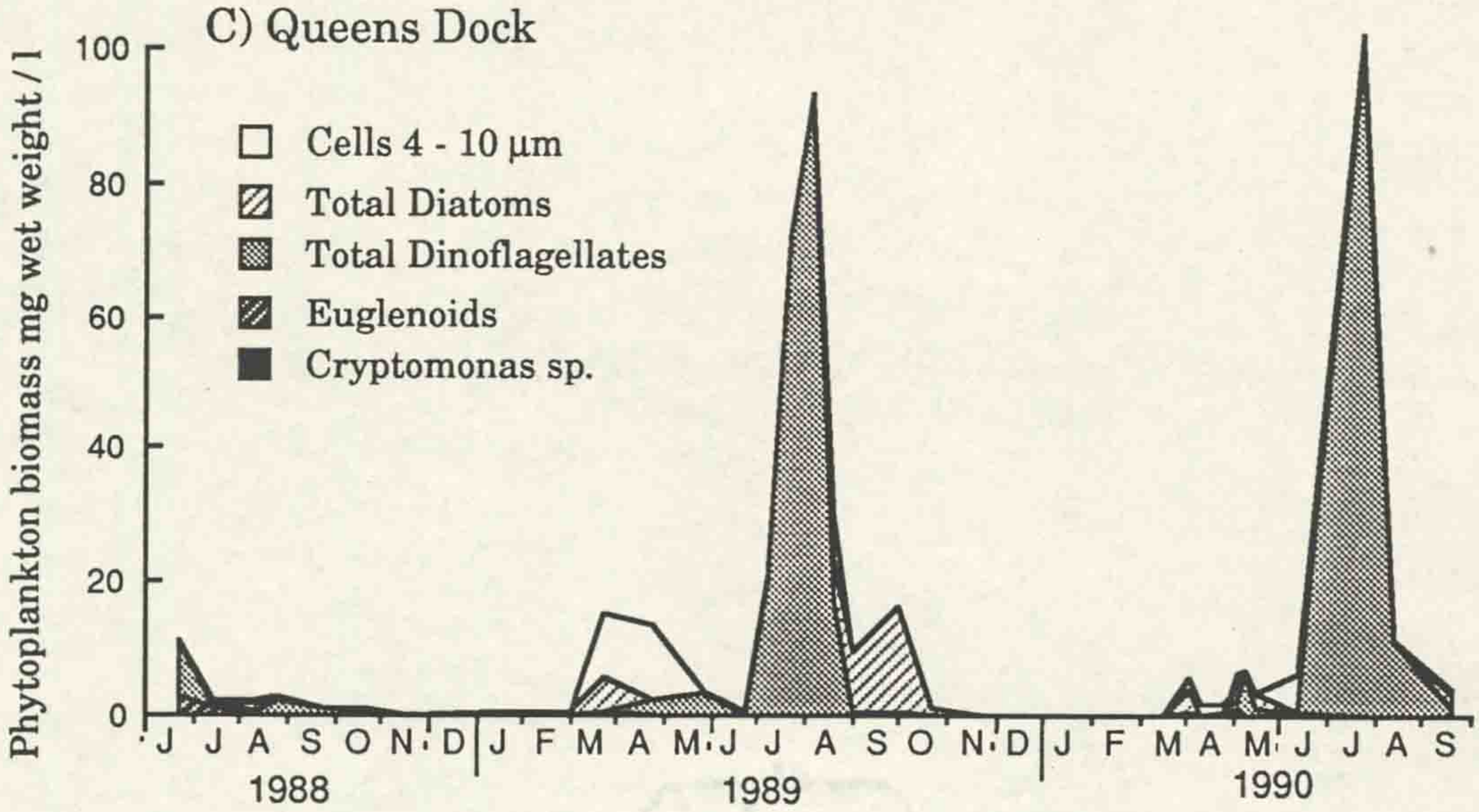


Fig. 4.2 Phytoplankton biomass, Graving (A) and Albert (B) Docks, surface waters. June1988 to September 1990. Note differences in scale.



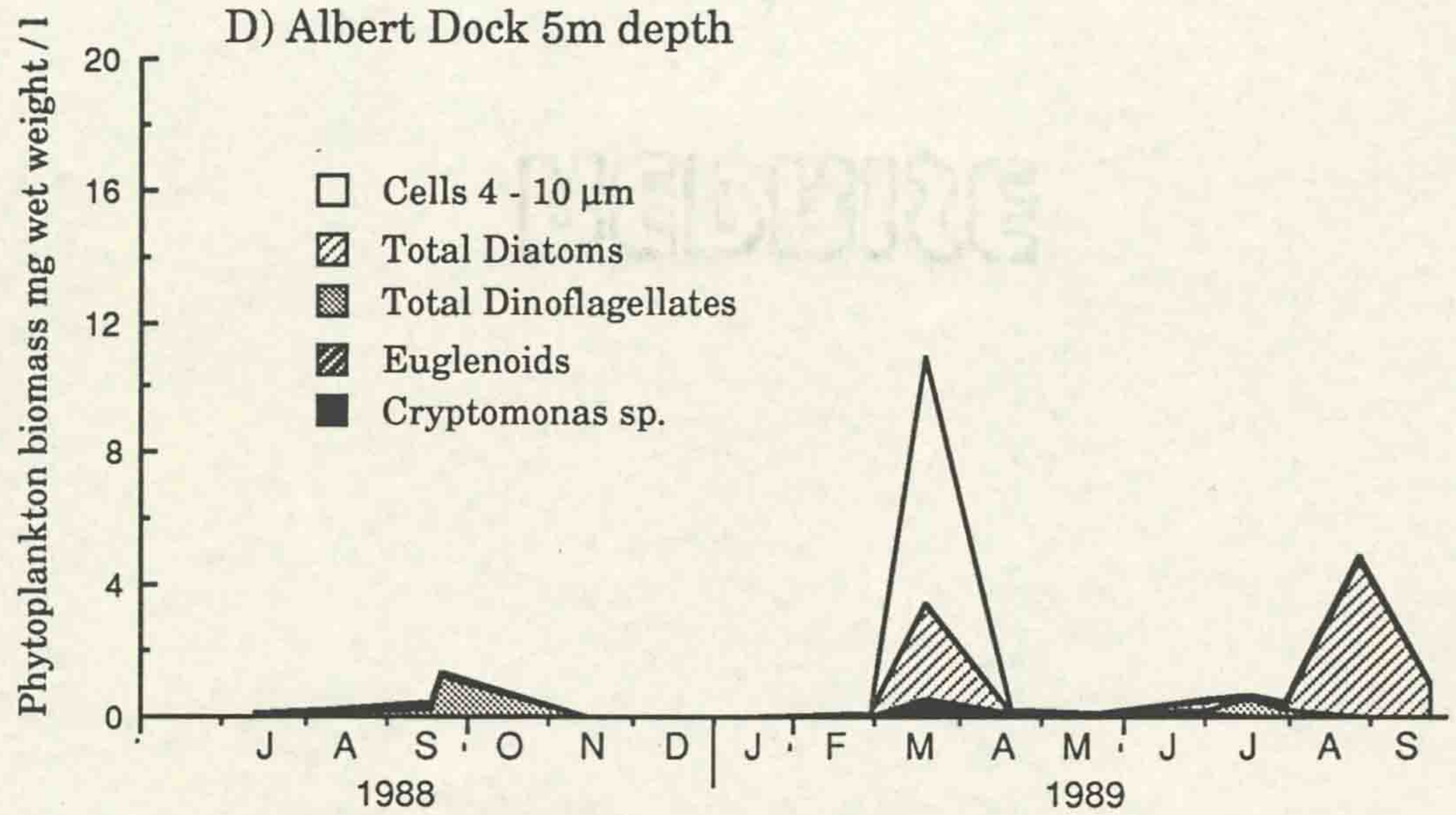


Fig. 4.2 C) Phytoplankton biomass, Queens suface water June 1988 to Setember 1990.
D) Albert5m depthJune 1988 to September 1989. Note differences in scale.

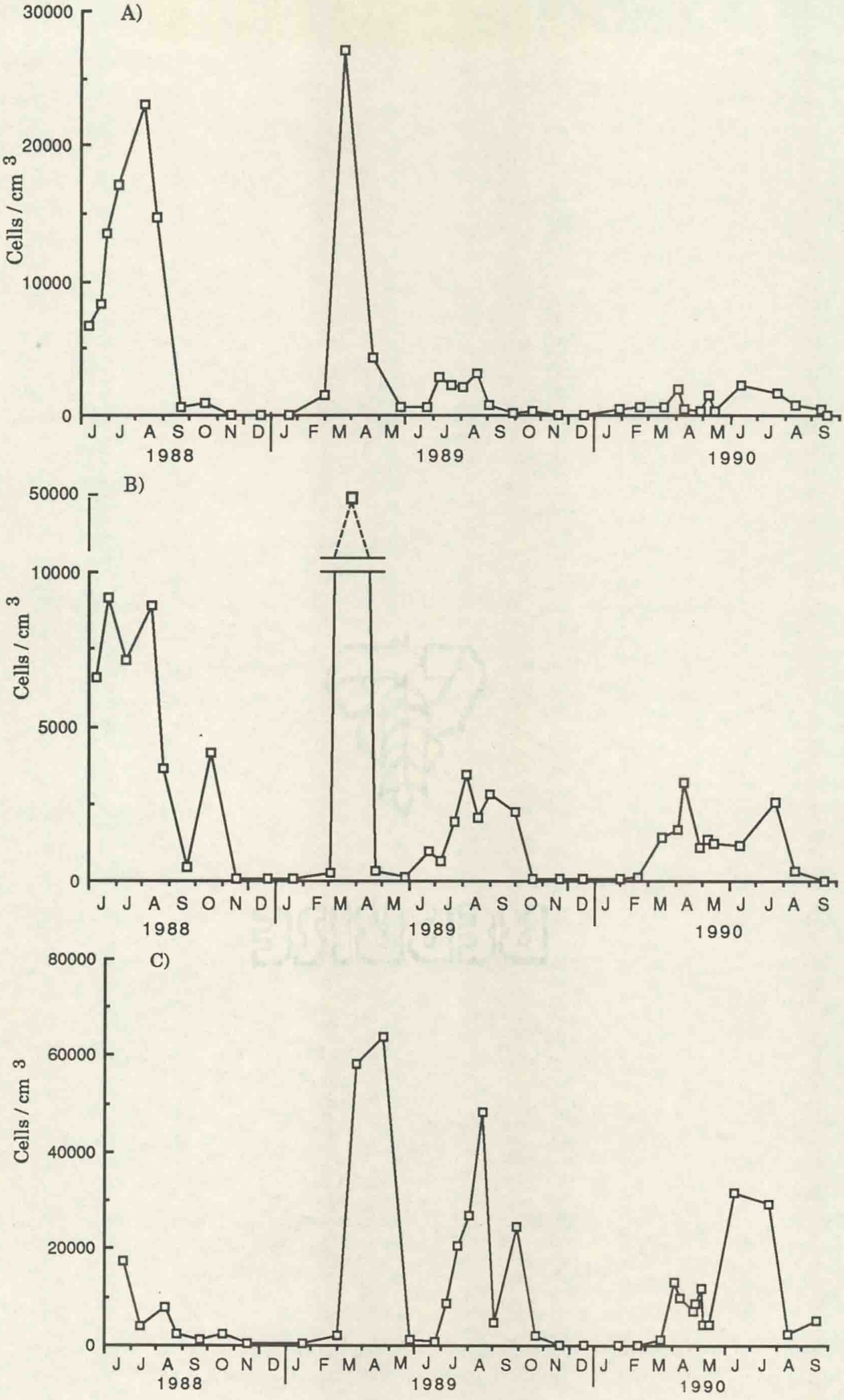


Fig 4.3 Total phytoplankton (> 4μm) cell densities in surface waters June 1988 to September 1990. Graving (A), Albert (B) and Queens (C) docks. Note differences in scale.

At the start of the sampling programme in June 1988 dense populations of dinoflagellates of the genus *Gymnodinium* (up to 13000 cells cm^{-3}) were present in all docks. In the Graving Dock, with the onset of artificial mixing, this was replaced by the euglenoid *Eutreptiella* which persisted for the rest of the summer. In the Albert and Queens Docks *Gymnodinium* populations also declined by mid July. In the Albert Dock this was followed by second biomass peak of the dinoflagellate species *Prorocentrum minimum* (up to 2500 cells cm^{-3}), but in the Queens Dock a comparatively low biomass of mixed dinoflagellate and diatom species followed, with *Cryptomonas* sp replacing the dinoflagellates in early autumn (Fig 4.2C).

Winter (November to February) phytoplankton populations were very low in all three docks (generally less than 100 cells cm^{-3}, Fig. 4.3). A dense spring bloom was recorded in March 1989 which was most marked in the Albert and Queens Docks, where total phytoplankton cell densities of up to 49000 and 62000 cells cm^{-3} respectively were observed. The small diatom species *Skeletonema costatum*, *Leptocylindrus danicus*, *Chaetoceros* spp and *Thalassiosira* sp and the colonial flagellate *Phaeocystis pouchettii* accounted for most of the total biomass in all docks at this time, although *Phaeocystis* was of greater significance in Queens and Graving than the Albert Dock.

An increase in dinoflagellate biomass (*Heterocapsa triquetra* and *Prorocentrum minimum*) occurred in the Queens and Graving Docks following the spring 1989 bloom (Figs 4.2 A,C). This was short lived, but by July *Prorocentrum minimum* dominated the phytoplankton of all three docks, being particularly dense in Queens Dock, where cell densities of up to 26,000 cm^{-3} were recorded (over 80 mgl^{-1} wet weight) Diatoms were once more the dominant phytoplankton group by September (Fig 4.2 A, B, C), the main species being *Leptocylindrus danicus*, *Skeletonema costatum* and *Thalassiosira* sp., with *Lithodesmium undulatum* in Albert Dock.

The peak spring phytoplankton biomass was much reduced in 1990 in all docks compared to 1989 (Fig 4.2 A,B,C). The component species were essentially the same, although *Phaeocystis* was a less important species than the previous year. A very dense bloom of *Prorocentrum minimum* again developed in Queens Dock in July 1990, but not in the Albert and Graving Docks. The phytoplankton biomass of Albert and Graving Docks was dominated by the diatom *Lithodesmium undulatum* in summer 1990 despite comparatively low cell concentrations (<500 cm^{-3}), because of the large cell volume of this species.

The large dinoflagellate *Noctiluca scintillans* was collected in zooplankton samples and is not included in the biomass or cell number figures for phytoplankton, results are given in appendix Tables IX to XI (as individuals m^{-3} as for all zooplankton samples). *Noctiluca scintillans* was observed in low cell densities (<20 individuals m^{-3}) in the Albert and Graving Docks in late summer 1989 and 1990, higher cell densities were recorded in the Queens Dock at these times (up to 530 individuals m^{-3}).

The total phytoplankton biomass of samples from 5m depth in the Albert Dock were much lower than surface samples (fig. 4.2A), apart from one occasion (September 1989) when diatom cells appeared to be concentrated close to the dock bottom. Species composition and succession were comparable in both surface and bottom waters, although the summer dinoflagellate peak tended to occur later in samples from 5m depth, possibly due to sinking of cells before demise of the bloom. The distribution of phytoplankton in the Graving Dock with depth is discussed in chapter 6 in relation to the effects of mixing.

4.3.2.2 Short Term Intensive Survey - Queens Dock

The more detailed study of the biomass (Fig 4.4 a) and cell concentrations (Fig 4.4b) in Queens Dock followed the species succession from the onset of the spring increase in primary production (early March) to the end of the first dinoflagellate bloom (mid May). A rapid progression from mixed diatom to *Phaeocystis* to dinoflagellate (*Heterocapsa*) dominated communities was clearly seen. The biomass maxima for *Heterocapsa* far exceeded the earlier

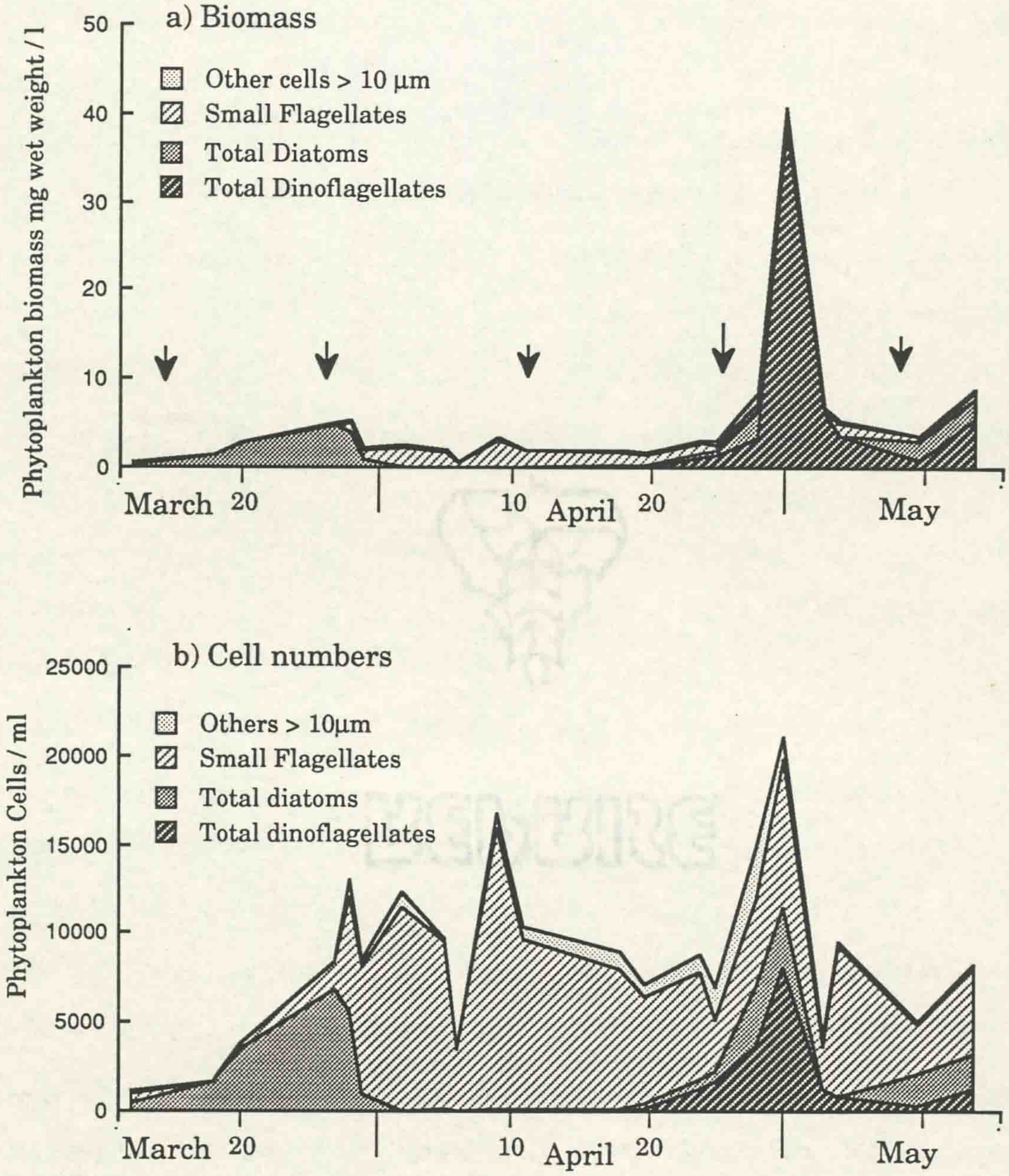


Fig. 4.4 Phytoplankton a) biomass, b) cell numbers, in Queens Dock surface water, 13th March to 14th May 1990. Arrows indicate water intake to dock.

peaks for diatoms or *Phaeocystis*, although the main dinoflagellate bloom (*Prorocentrum minimum*) was not until July. Comparison between cell biomass and cell number graphs shows that cell counts alone can give a misleading representation of the importance of numerous but small species such as *Phaeocystis* .

Intake of water from the Mersey into the dock did not result in major changes in phytoplankton numbers or biomass during the diatom/*Phaeocystis* phases. The *Heterocapsa* peak, however, occurred just after a topping up event and declined prior to the next (a period of twelve days).

Planktonic protozoa and rotifers were also enumerated in the phytoplankton samples. Planktonic protozoa (mainly oligotrichs) reached peak numbers at the end of April, coinciding with the *Heterocapsa* bloom. Rotifers were only present in large numbers on the 9th and 11th of April, coinciding with the *Phaeocystis* peak, raising the possibility that this was their major food source.

4.3.3 Zooplankton

Similar trends in zooplankton numbers occurred in all three docks with the very high maxima of summer 1988 dropping by three orders of magnitude to the maxima of summer 1989 and 1990 (Fig. 4.5). These observations are confused to some extent by the fact that a finer mesh net (140μm rather than 250μm) was used for sampling in June and July 1988. Samples taken using a 250μm net during this same period by Mincher (1988), however, showed numbers of the same order of magnitude as those obtained using the 140 μm net (Fig 4.5). Temporal variation in three of the main zooplankton groups (copepods, barnacle nauplii and medusae) are illustrated in Fig. 4.6 (densities of individual species for each sampling date are given in appendix Tables IX to XI). The dense zooplankton communities of 1988 consisted mainly of the polychaete larva *Polydora ciliata*, the calanoid copepod *Eurytemora affinis* and additionally, in Queens Dock the cladoceran *Podon*. Copepods of the genus *Acartia* (*A. clausii* and *A. bifilosa*) and barnacle nauplii occurred in lower numbers. No *Eurytemora* was found in 1989 and the

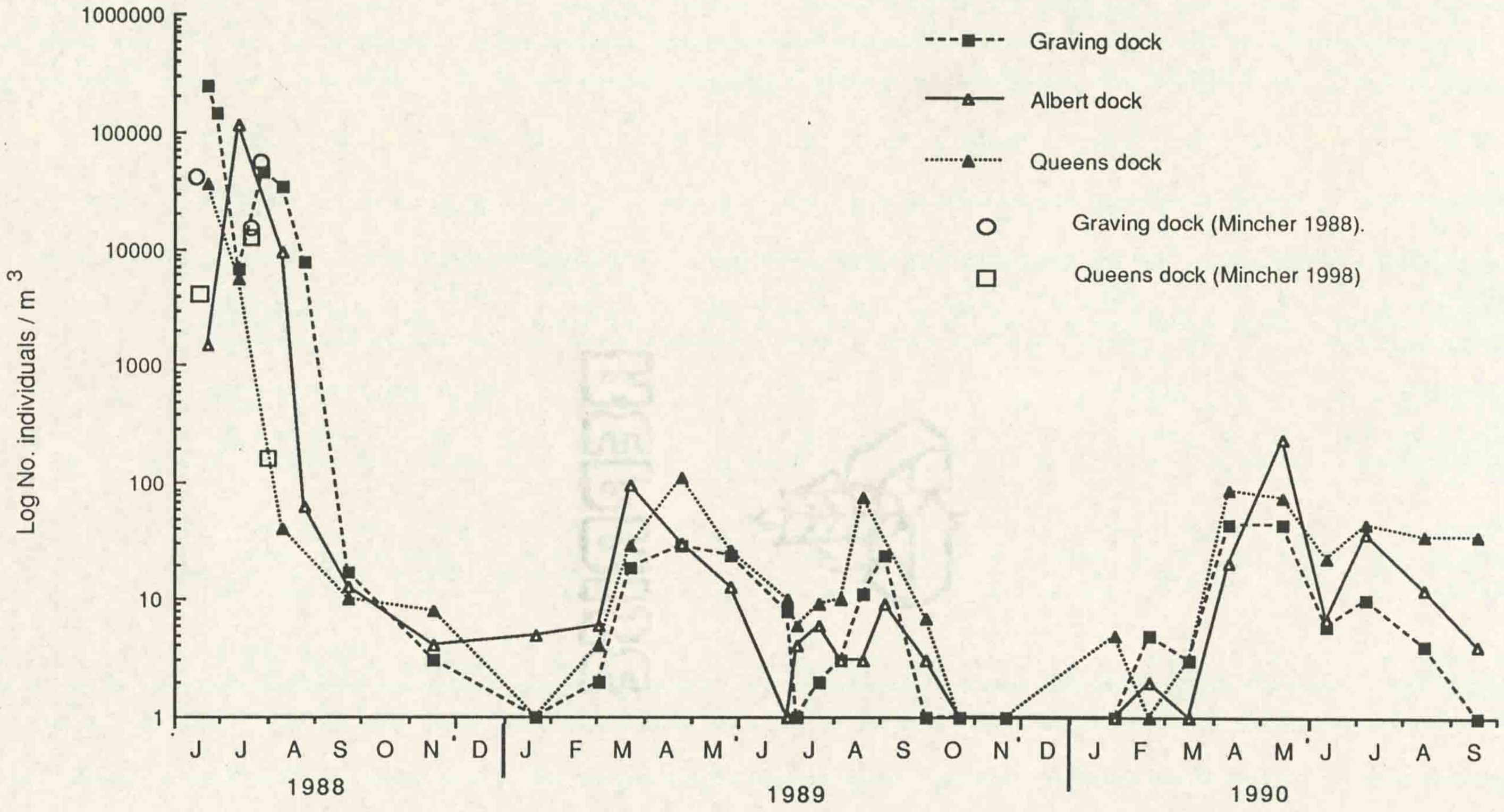


Fig 4.5 Total concentrations of zooplankton in the Graving, Albert and Queens Docks, June 1988 to Sept. 1990. June / July 1988 samples taken with 140µm mesh net, all other samples taken with 250µm net. Samples taken in June / July 1988 using a 250µm net by Mincher (1988) are shown for comparison.

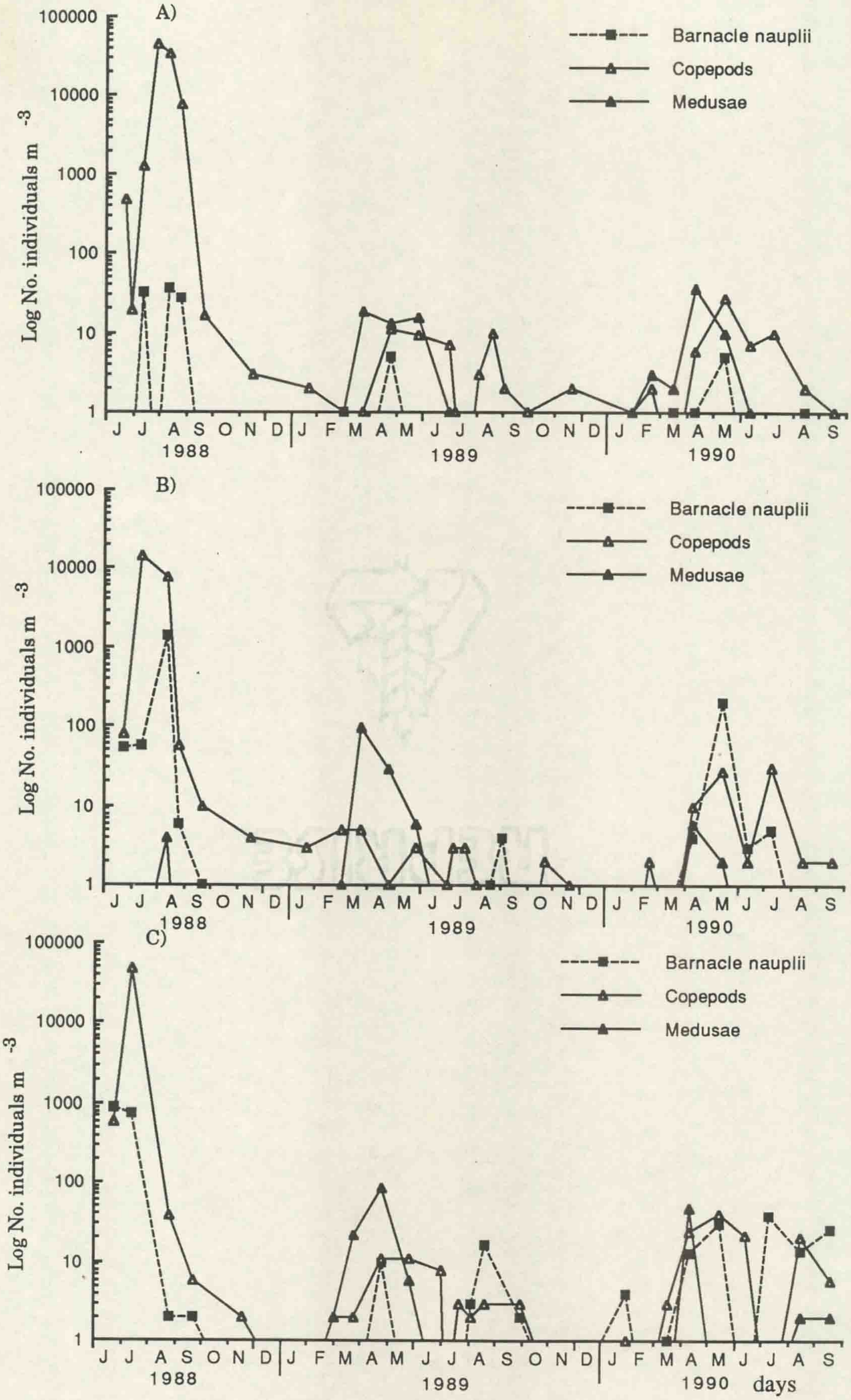


Fig. 4.6 Densities of Barnacle nauplii, Copepods and Medusae in the Graving (A), Albert (B) and Queens (C) Docks, June 1988 to September 1990.

numbers of *Polydora* and *Podon* were low in both 1989 and 1990. The spring - summer zooplankton populations in 1989 and 1990 were at densities of 10 to 100 animals m^{-3}, while winter populations fell to below 5 animals m^{-3}, often with no animals found in samples.

Throughout 1989 and 1990 zooplankton communities typically consisted of low populations of the copepods *Tisbe longicornis* (Scott), *Tigriopus brevicornis* (Müller), *Acartia* sp or *Eurytemora affinis*, with the occasional presence of ascidian, anthozoan, cyprid and polychaete larval stages and *Carcinus maenas* zooea. Medusae stages of the cnidarians *Obelia* and *Aurelia aurita* were present in late spring. Small specimens of *Aurelia aurita* were abundant in April. Larger specimens were evident in large numbers in all docks later in the year and were occasionally caught in the zooplankton net, as was the ctenophore *Pleurobrachia pileus*, however the method of sampling employed did not favour the capture of such large species.

A small but persistent community of microzooplankton (naked ciliates and tintinnids) was present in all docks throughout the year (appendix Tables VI to VIII) and were enumerated in phytoplankton samples due to their small size. Maximum abundance occurred from April to September.

4.4 DISCUSSION

4.4.1 Plankton Ecology

If the temporal variation in chlorophyll *a* and phytoplankton biomass for Queens Dock are compared it is obvious that chlorophyll *a* was not always a good indicator of phytoplankton biomass. During dinoflagellate blooms, chlorophyll *a* concentrations often remained relatively low despite very high dinoflagellate biomass. Such under-estimation of phytoplankton biomass or productivity where dinoflagellates are concerned has been reported for other systems (e.g. Hickel *et al* 1971, Gieskes & Kraay 1975).

Other results highlight important considerations for sampling strategy. The detailed short term study of phytoplankton in Queens Dock illustrated that large changes in phytoplankton numbers can occur over periods of only a few days. Hence the sampling frequency adopted of once or twice monthly gives little detail of successional patterns. As seasonal changes are over several orders of magnitude, however, the main trends are apparent. Sampling of phytoplankton and chlorophyll *a* at different depths in the Albert and Graving Docks showed marked vertical heter0geneity in concentrations and species. This is due to a combination of factors including sinking effects and, in the case of dinoflagellates, phototactic vertical migration (Eppley *et al* 1968). Hence if phytoplankton populations in the dock as a whole are to be effectively described, either detailed depth samples or integrated sampling of the whole water column is required.

The dominant phytoplankton species in the South Docks are typical of nutrient rich, polyhaline or coastal waters. As in the South Docks, *Skeletonema costatum* and *Phaeocystis pouchettii* are the main spring bloom forming species in the lower Mersey estuary (Sharples 1972, G. Russell unpub., Jemmett pers. comm.). However, the main nuisance species in the docks, *Prorocentrum minimum* and to a lesser extent *Heterocapsa triquetra*, are not reported to form dense populations in the estuary or surrounding coastal waters. The lack of water movement in the South Docks with associated physical stability, and the relatively high water temperatures in summer stimulates the growth of some dinoflagellate species (Hickel *et al* 1971, Holligan *et al* 1980). The large dinoflagellate *Noctiluca scintillans*, a relatively unimportant dinoflagellate in the South Docks, is known to form dense aggregations in Liverpool Bay (Burrows 1975).

Temporal variation in both chlorophyll *a* and phytoplankton showed marked annual variation with peak production in March or July and very little activity in winter. This is typical of many temperate marine and freshwater systems. It is not always the case, however, in disused docks where substantial phytoplankton biomass may be present throughout the winter. High winter phytoplankton biomass has been observed in Salford Quays (K. Hendry

pers. comm.) and Preston Docks (Conlan 1989). The phytoplankton concerned were blue-green algae, which are capable of growing at the low light intensities of winter but are uncommon in more saline waters such as the South Docks.

The cause of the substantial decline in maximum spring diatom / *Phaeocystis* biomass between 1989 and 1990 in all three docks is unclear. The trend continued in 1991 with spring phytoplankton numbers generally even lower than in 1990 (J. Eaton pers. comm.). Nutrient levels, zooplankton populations, water temperature, and hours of sunshine were similar in both years and do not appear to explain the different spring patterns. Although differences in phytoplankton biomass were seen between 1989 and 1990 the species progression from spring to summer was very similar. The dominant spring diatom *Skeletonema costatum* is typical of polyhaline, moderately polluted waters (Umamaheswara Rao & Monchand 1988). Demise of the spring diatom population is almost certainly due to silicate limitation, silicate levels falling rapidly from the onset of diatom growth in late winter. *Phaeocystis* is able to grow rapidly following the diatom bloom as it does not require silicate for growth. This species is often present in very high numbers in coastal waters in late spring (Sharples 1972, Bätje 1986).

Scripsiella sp. and *Heterocapsa triquetra* are the first dinoflagellates to appear after the spring diatom bloom. These species are typical of mixed waters (Holligan *et al* 1980). Dense populations of *P. minimum*, as seen in Queens Dock, were also reported for Obidos Lagoon following an earlier *S. costatum* bloom (Silva 1985). The growth of *P. minimum* in this lagoon was associated with physical stability of the water and high nutrient concentrations. Hence the succession of dinoflagellate species seen throughout the season is associated with changes in both temperature and stability of the water. It was also suggested that the *P. minimum* blooms thrived in Obidos Lagoon due to high levels of organic nutrients supplied by the decaying diatoms. In the South Docks *P. minimum* blooms are associated with the maximum annual water temperatures from the end of June to September when stability of the water is also likely to be highest. A supply of organic nutrients may be provided by the

decaying spring algal populations or by inputs of estuarine water during topping-up. The formation and maintenance of such dinoflagellate blooms may be achieved by the suppression of zooplankton grazing (Fielder 1982, Huntley 1982). It is possible that the same effect occurs in some benthic filter feeders, the filtration rate of *Mytilus edulis* has been found to be reduced during a dinoflagellate bloom (Widdows *et al* 1979).

Remarkable differences in the scale of the *P. minimum* populations between docks were observed in 1989 and 1990, with very high biomass in Queens Dock compared to the Albert and Graving Docks. This is probably due to a combination of factors namely: the more direct input of estuarine water rich in organic nutrients, the shallow depth of the water body, less benthic grazing pressure and less shading effect by dock buildings and walls, which will all favour growth of algae in the Queens Dock.

The planktonic flora and fauna present in the South Docks may originate from populations resident in the dock or be introduced with water from the Mersey estuary during 'topping up' operations. Introductions of species may provide the seed for the development of phytoplankton and zooplankton blooms either by direct provision of mature reproductive stages in the case of phytoplankton and holozooplankton or, in the case of merozooplankton, by providing the larval supply for the development of a benthic fauna which may release further larval stages directly into the dock at a later stage. Introduced species will always be those typical of the high salinity regions of the Mersey Estuary as water intake is always carried out at high tide.

The zooplankton of the South Docks is typically estuarine in character. The dominant copepods of summer 1988, *Eurytemora affinis* and *Acartia* spp are also dominant in many estuaries of the British Isles. *Eurytemora affinis* is typically found in the higher estuarine reaches of salinities less than 30 °/oo while a complex of *Acartia* spp is generally found in the lower to middle reaches of salinities 27 - 33.5 °/oo (Collins & Williams 1982, Taylor 1987). These copepods are present in large numbers in the Mersey Estuary throughout the year (A. Jemmett pers. comm.). Both are known to be able to utilise detrital material as a food source

when phytoplankton populations are low (Heinle 1977). A preliminary survey of the summer zooplankton of the Mersey Estuary (Wilkinson *et al* 1990) found *Polydora* as one of a number of polychaete larvae. The same survey however, found gastropod and lamellibranch larvae to be of far greater importance in the Mersey than in the docks. The sparsity of lamellibranch larvae in zooplankton samples, specifically of *Mytilus edulis*, is anomalous as very dense settlement of mussels occurred throughout the South Docks during the sampling period and the mesh size of the net should have been suitable. Some temporal or spatial deficiency in sampling strategy must have occurred but the precise nature of this is unclear.

The most notable variation in zooplankton populations over the study period is the dramatic change in population structure and densities between summer 1988 and 1989/1990. A decrease in the densities of zooplankton such as that seen could be due to several factors: increased predation, reduced food supply, deterioration or unsuitability of environmental conditions and, in the case of meroplankton, other factors affecting the source of supply .

It is unlikely that environmental conditions could directly explain reductions in zooplankton numbers as water quality has generally improved and physico-chemical parameters such as salinity, temperature and pH showed no major differences between summers. The presence of large numbers of dinoflagellates could adversely effect copepod production as some dinoflagellate species have been shown to reduce feeding activity, egg production and survival and this may effect population dynamics if blooms persist for the generation time of the copepods (generally around 2 - 3 weeks) (Gill & Harris 1987). However, this could only be a factor in the Queens Dock as this is the only site where dinoflagellate populations were greatly increased in summer 1989 and 1990.

A reduced phytoplanktonic food supply could have resulted in lowering of zooplankton production in the Albert and Graving Docks in 1989 and 1990: as discussed in chapter 6 this may be due to competition for food by benthic filter feeders.

Increased predation on zooplankton has almost certainly taken place over the period of decline in zooplankton production. This controlling factor would apply to all docks and all species of macrozooplankton and as reductions in populations have occurred in this way this would seem the most likely explanation. No routine studies of fish populations were carried out as part of this study, but various other short term investigations have been carried out. Mincher (1989) found fish populations to be very low in initial gill nettings in the Graving and Queens Docks in summer 1988. However, later investigations using the same methods by Heaps (1990) and Lonsdale (1990) found a large increase in fish populations including planktivorous fish such as sprats. Feeding by planktivorous fish has been shown to reduce numbers of large-bodied zooplankton (Brooks & Dodson 1965, Lynch 1979, Fulton 1984) and this may have happened in the South Docks. Other predators may also have contributed to the decline. Populations of *Pleurobrachia pileus* and *Aurelia aurita* were present in the docks in all summers studied but the relative numbers between years is not known. Both species feed on zooplankton and may have affected populations. Ctenophores are known to strongly affect estuarine copepod populations (Lonsdale 1981). Predation of the smaller larval stages of zooplankton by benthic filter feeders is another possibility. Ascidians have been shown to filter out planktonic larvae (Young 1989) but little information is available on other species combinations or sizes effected.

The presence of very high densities of *Polydora ciliata* in summer 1988 and their subsequent dramatic decline in the following summers may be due to successional changes in the benthos present in the docks over this period. Benthic habitats in the South Docks, shortly after reflooding with water, were sparsely populated and these resources were quickly exploited by settling larvae introduced with estuarine water. Once a breeding population was established any larvae released into the dock would be confined there. As early colonising, opportunistic species are usually r - strategists it might be expected that large numbers of the larvae of these species would be released and confined in the dock waters. As succession of the dock benthos progressed and opportunistic species were replaced by a diversity of other fauna the numbers of early colonising larval stages might be expected to decrease.

In Sandon Dock a similar reduction in zooplankton production was also seen sometime after limitation of water exchange, although reasons for this were not studied (Hawkins , pers comm.).

Microzooplankton may be an important component of marine food chains, consuming up to 70 % of production in some studies (Burkhill 1982), but are sometimes completely overlooked. A large proportion of the food of such protozoa is likely to be nanoplankton (Spittler 1973, Burkhill 1987), although bacteria and detritus may also play a part (Sorokin 1981, Hollibaugh *et al* 1980).

4.4.2 Implications For Water Quality

Nuisance phytoplankton populations are directly responsible for some of the greatest shortfalls in water quality in the South Docks. Low water movements and redirected run off result in low levels of suspended solids, so poor water clarity is directly attributable to phytoplankton concentrations. In the summer months phytoplankton may cause the water to be strongly coloured, particularly during dinoflagellate blooms, which may turn the water a bright orange-brown. Dense algal populations cause reductions in dissolved oxygen levels when in decay and it is likely that this is a major factor in the oxygen deficiencies in the South Docks.

Several of the dinoflagellate species present in the docks are potentially toxic species. The most prevalent and potentially the most toxic is *Prorocentrum minimum*. *Prorocentrum minimum* has been responsible for outbreaks of both paralytic and diarrhetic shellfish poisoning in many areas of the world (Freudenthal & Jijna 1979, Kat 1979, Silva 1985). *Noctiluca scintillans* has been shown to produce toxins but the toxic effects of blooms appears to be low (Morton & Twentyman 1971). Other dinoflagellates identified only to genus due to low numbers or difficulty of preservation are from genera known to include toxic species. The

most notable of these are *Dinophysis*, which has been responsible for cases of diarrhetic shellfish poisoning (Russell 1984 , Kat 1987) and *Gymnodinium* which has been responsible for mortality of marine organisms (Vagn Hansen & Sarma 1969).

Although no direct toxic effects of phytoplankton have been observed in the South Docks on either marine fauna or humans the possibility is a cause of concern to the dock management authorities and has occasionally resulted in the closure of water sports facilities as a precautionary measure. The presence of dinoflagellates possessing potentially lethal toxins would be a serious deterrent to the establishment of a commercial mussel culture business, similar to Sandon Dock, in any dock with similar problems.

Large *Phaeocystis* populations may cause the formation of a thick foam scum in coastal waters when combined with high wave activity (Bätje 1986): this is unlikely to be a problem in the South Docks where turbulence is limited by the sheltering action of the high dock walls.

The assumption that improvements in water clarity in the Albert and Graving Docks in summer 1989 and 1990 compared to 1988 was due to lower phytoplankton populations is confirmed by a corresponding reduction in phytoplankton biomass and chlorophyll *a* concentrations. Water clarity may also be affected by the plankton species present, smaller particle sizes cause greater turbidity (Postma 1961), hence a shift from small to larger cells may result in an increase in water clarity (Bakker & De Pauw 1974). Such a change was seen in the Albert and Graving Docks with a decline in small flagellates and the appearance of the larger diatom *Lithodesmium undulatum*.

The densities of both macro and microzooplankton may have a direct impact on water quality through the control of phytoplankton populations by grazing. Thus the decline in zooplankton which occurred in the South Docks may be seen as undesirable in terms of water quality

management. The maintenance of high numbers of planktonic grazers has been used in recent years as a biomanipulation tool for improving water quality in fresh waters; this is discussed further in later chapters.

In summary, the plankton of the South Docks is of lower estuarine character with several species being typical of eutrophic or organically enriched waters. Phytoplankton blooms are a major water quality problem, both in terms of the visual appearance of the water and the presence of potentially toxic species. These problems were found to be reduced in the Albert and Graving Docks in summers 1989 and 1990 compared to 1988, while no such improvement was observed in Queens Dock. The most notable temporal variation in zooplankton communities was a dramatic decline in numbers in summers 1989 and 1990 when compared to 1988.

CHAPTER FIVE

PRELIMINARY OBSERVATIONS ON BENTHOS AND FISH

5.1 INTRODUCTION

The marked changes in environmental conditions which have taken place in the South Docks over the last ten years open many interesting opportunities for the study of marine/estuarine macroflora and macrofauna. After being silted over, dredged and then refilled with water these docks provided a 'clean slate' for colonisation, allowing the study of successional patterns in a relatively controlled and stable environment. The study of the macrobiota is an important aspect in the understanding of water quality, both in terms of presence of indicator communities and possible modification of water quality by resident species. A study of the communities present in the South Docks was also desirable in the light of considerations of the value of such an urban waterbody as a resource for conservation and education.

A preliminary investigation of the sediment fauna of the South Docks carried out for the Merseyside Development Corporation in early 1988 (Altwell 1988) found sparse populations of four species of polychaetes in diver collected cores, mainly *Capitella capitata* . A later study found no polychaetes in grab samples (Mincher 1988).

Russell *et al* (1983) described the flora and fauna of the dock walls of two Merseyside docks, Sandon Dock, used as a shellfish farm and Brocklebank Dock, used for commercial shipping. Surprisingly diverse communities were present in both docks, the greatest diversity being found in Sandon Dock where water quality was improved by aeration. The sediment fauna of another polyhaline Merseyside dock, Collingwood Dock, was studied by James and Gibson (1980) who found 13 species, all polychaetes or crustaceans, *Capitella capitata* being numerically dominant.

Previous work on the macrobiota of docks has been carried out in several other areas in the U.K. Brief descriptions of the flora and fauna found on dock walls and in sediments are given in Hendry *et al* (1988a) for ten docks of varying salinity around the U.K., with a more

intensive study in Preston Dock being carried out by Conlon (1989). No sediment dwelling benthos was found in these studies although a benthic macrophyte community (consisting of rooted angiosperms) was present in the firmer substrate of Cavendish Dock at Barrow - In-Furnesss. The diversity of biota on the dock walls varied considerably with salinity and water quality in the docks surveyed.

Short- term investigations of the fish in the South Docks have been carried out (Mincher 1988, Heaps 1989, Lonsdale 1990). Information on the fish populations of several other docks is also available (Conlon *et al* 1988, 1989, Hendry *et al* 1988a, b,). The diversity of fish populations was often surprisingly high, even in docks with relatively poor water quality, a maximum of 20 species being found in one dock (Floating Harbour, Bristol).

In terrestrial systems the sequence of colonisation of 'bare ground' is well documented (see Krebs 1978 for overview). The picture is less clear for marine benthic communities, particularly of subtidal hard substrates. In marine systems the sequence of colonisation and community development is subject to a wide range of variables. The species involved in colonisation are determined to a large extent by propagule supply which can be strongly influenced by chance factors (Keough 1983, Underwood & Fairweather 1989). Other important factors include larval behaviour (Keough & Downes 1982, Crisp 1984), the nature of the substrata (McGuiness 1989, Richmond and Seed 1991) and biological interactions (Connell & Slatyer 1977). In established communities species composition and abundance is also influenced by physical or biological disturbance and competition (Paine 1966, Dayton 1971, Osman 1977, Sutherland 1974, Murray & Littler 1978).

Various models for succession have been proposed (see Krebs 1978, Valiela 1984, Richmond & Seed 1991 for reviews). The earliest models put forward the theory that early colonists prepare the way for later arrivals and were based initially on observations on terrestrial plant communities (Warming 1909, Cowles 1901, Clements 1936, Scheer 1945). However, successional interactions range from the essential to the inhibitory and this led Connell &

Slatyer (1977) to propose three models: 'facilitation' whereby resident species enhance the establishment of invading species; 'tolerance' where early colonisers have no effect on later arrivals; and 'inhibition' where species resist invasion so that longer lived species remain dominant. More recently the influence of stochastic effects, particularly due to unpredictable larval supply has been recognised as a major factor shaping community development (Fager 1971, Svane 1988, Underwood & Fairweather 1989). Several mechanisms of succession may affect the development of a community and the prediction of a stable end point is difficult, if indeed such a point exists at all (Richmond & Seed 1991).

The initial stages of colonisation of soft sediments after organic enrichment disturbance is reviewed by Pearson & Rosenberg (1978). Initial colonisation by *Capitella capitata* followed by Spionidae (usually *Polydora* spp) was found to be common to many studies. The dominance of organisms such as *Capitella capitata* in polluted substrates may not simply be due to better stress tolerance abilities than competitors but also to opportunistic advantages of life history and methods of reproduction (Gray1981).

In freshwater, pollution indices based on the presence of faunal indicators are a widely accepted method of assessing water quality. Attempts have been made to develop a similar index for marine benthic communties (e.g. Reish 1972, Bellan & Bellan-Santini 1972), but the greater variety of marine environmental conditions and species prevents the development of a widely applicable system (Pearson & Rosenberg 1978).

Direct effects of macrobiota on water quality act mainly through the control of phytoplankton blooms, either by increasing grazing pressure or reducing nutrient concentrations (e.g. King 1980, Officer *et al* 1982, Dorazio *et al* 1987). Such interactions are of great interest as water quality management tools and are considered in greater detail in chapters 6 and 7.

The flora and fauna of the South Docks was studied from May 1988 to Janaury 1991 in the Graving, Albert and Queens Docks. A detailed study of the macroflora and macrofauna of

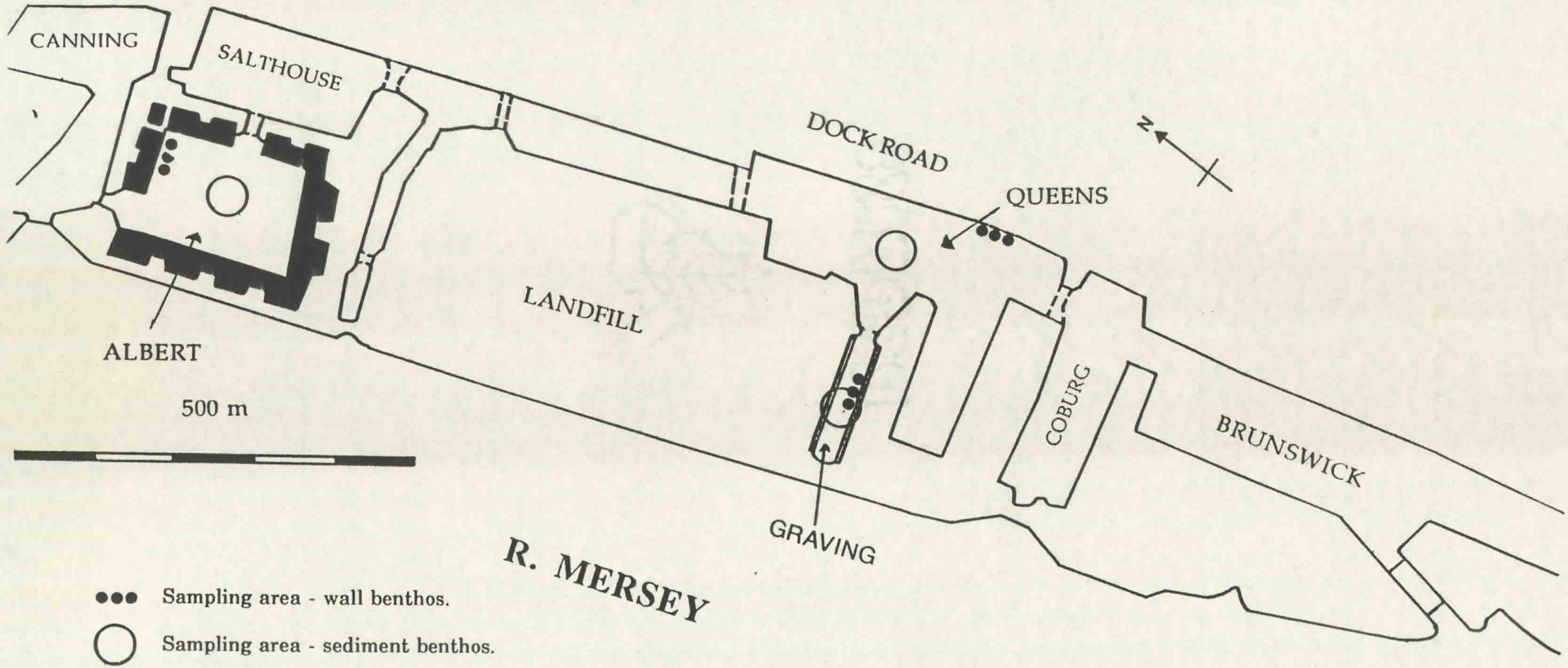


Fig. 5.1 Map of South Docks showing sampling areas for wall and sediment dwelling benthos.

the South Docks was beyond the scope of this research project, which was directed mainly at water quality management. This chapter summarizes various preliminary observations on the ecology of benthos and fish. These included a semiquantitative study of the diversity and distribution of marine life in the Graving, Albert and Queens Docks, in order to record the main community types and any major temporal or spatial variations. Some very preliminary work was carried out on other Merseyside docks. This work has been included in this thesis, despite its limitations, to clarify any possible interactions between macrobiota and water quality. More detailed studies on the population dynamics of *Mytilus edulis* were carried out because of their possible importance as a biological filter, and their dominant role in the communities of the dock walls.

5.2 METHODS

5.2.1 Wall Communities

The abundance of species covering the walls or forming an algal canopy was assessed as percentage cover in the Graving, Albert and Queens Docks, along the walls at sites indicated in fig 5.1, by diving survey. Subjective estimations of percentage cover were made, using point quadrats (0.25 m^2 16 point quadrats) as a guide in difficult cases, at 1m depth intervals from surface to bottom, on vertical wall surfaces. Estimations were carried out at three stations at each site, chosen from the surface without prior inspection below water. The average percentage from the three stations was taken for each depth.

Destructive sampling was used to collect smaller species and epifauna for identification and enumeration. A 25 x 25 cm quadrat was scraped into a muslin bag from each depth at one station only. On returning to the laboratory the bags were washed in a 1mm mesh seive to separate mussels from other flora and fauna. All species present were noted and smaller faunal species and epifauna on mussels were counted.

Diving surveys for such abundance estimations and destructive sampling were carried out in June 1988 (Queens and Graving Docks only), July 1989, December 1989, May/June 1990, August 1990 and January 1991 (Albert Dock only).

5.2.2 Timed Searches

Five minute timed searches were carried out by divers in the Graving Albert and Queens Docks, at the same locations as for wall sampling (Fig 5.1), in order to record the presence of infrequent or mobile species missed by other methods of sampling. Searches were confined to the the dock wall and up to 1m onto the sediment in the Albert and Queens Docks, and in the Graving Dock to the top 4m of wall, including the horizontal substrate of the steps. Species recorded were added to the combined species list (Table 5.1). Detailed results are included in appendix Table XVIII.

5.2.3 Sediment Fauna

Sediment samples were gathered using a 0.09m^2 Van Veen type grab in the areas indicated in fig 5.1. Samples were collected in August 1989 and August 1990 for Graving, Albert and Queens Docks and in August 1989 only for Brunswick Dock. Three grabs were taken from each dock. These were pooled and seived through a 1mm mesh sieve. Fauna retained was identified and enumerated. Due to the small number of samples and limited penetration of such grabs it should be noted that results indicate only very approximate densities of the more abundant animals from upper layers of the sediment.

5.2.4 Fish

No routine sampling of fish populations was undertaken as part of this study, but information was gathered by a number of methods. In the summers of 1989 and 1990 live fish were needed for public exhibitions of the marine life of the docks. Fyke nets and small bottle fish traps

were placed in the Albert and SalthouseDocks for a 3 to 6 week period during May to June of these years and fish removed twice weekly. Line fishing with barbless hooks was also carried out during these periods. Fish collected by these methods were identified and are included in the species list. Other records originate from *ad hoc* observations made while SCUBA diving for work or pleasure.

Gill netting of the South Docks was carried out periodically by colleagues from Manchester University as part of a broader programme of dock ecology. Any other species of fish caught in this way were included in the combined species lists.

5.2.5 Combined species list (Table 5.1 a, b)

This species list was compiled using records from the above methods, collections of unusual material observed during sampling trips and from animals gathered by teams of divers for the 'Dockwatch' and Living Waters' exhibitions.

5.2.6 Other Docks

The fauna of five other Merseyside docks were sampled as part of a survey on behalf of the Liverpool Museum. Long-handled scraper nets of two mesh sizes were used to gather biota from approximately 0.5m depth. Five replicate areas, 0.25 x 0.25 m were scraped with the larger, course mesh net for the collection of macrobiota. A smaller fine meshed net was used for the collection of smaller organisms, five replicate scrapes, 0.2 x 0.08 m^2 were taken. These nets did not necessarily remove 100 % of the biota from within these areas.

5.2.7 Mussel Populations

Mussels were collected from Albert, Queens and Graving Docks concurrently with sampling of wall benthos in August 1989, November 1989, May 1990, August 1990 and January 1991 (Albert Dock only). Three replicate 25 x25 cm quadrats were scraped into mesh bags at 1m

depth intervals at three stations for each site. The total whole wet weight and number of mussels in each sample was determined.

The three replicate scrape samples from 1m depth were then pooled. The lengths of 200 randomly chosen individuals (less in initial samples or those with insufficient numbers) were measured to give the population structure. 50 individuals (occasionally less) from a wide range of sizes were selected for biometric measurements in which the whole wet weight (W.W.W.), shell weight (S.W.), dry weight of soft parts (D.W.S.P.) and length measured for each animal. The mean D.W.S.P. for the population was calculated by regression of Log n D.W.S.P. against Log n length. Conversion of length to D.W.S.P. was then carried out for the lengths of the 200 randomly selected animals and the mean D.W.S.P. calculated. No weight data is available for November 1989 due to freezer failure and consequent loss of stored specimens.

5.3 RESULTS

5.3.1 Wall Benthos

The abundance of main algal and faunal groups in the Graving, Albert and Queens Docks, with depth are illustrated in figs. 5.2 to 5.4 for several sampling occasions. Detailed information for all sample dates is given in appendix Tables XII to XIV.

On initial examination of the walls in the Graving and Queens Docks in June 1988 the diversity of fauna and flora was very low. Bryozoa, probably mostly *Conopeum* spp. formed the dominant cover on the walls being abundant in the Queens Dock and more patchily in the Graving Dock. The fauna of the Graving Dock was less diverse than in Queens: no *Nereis* or amphipods were found although *Polydora* was present on the horizontal parts of the walls.

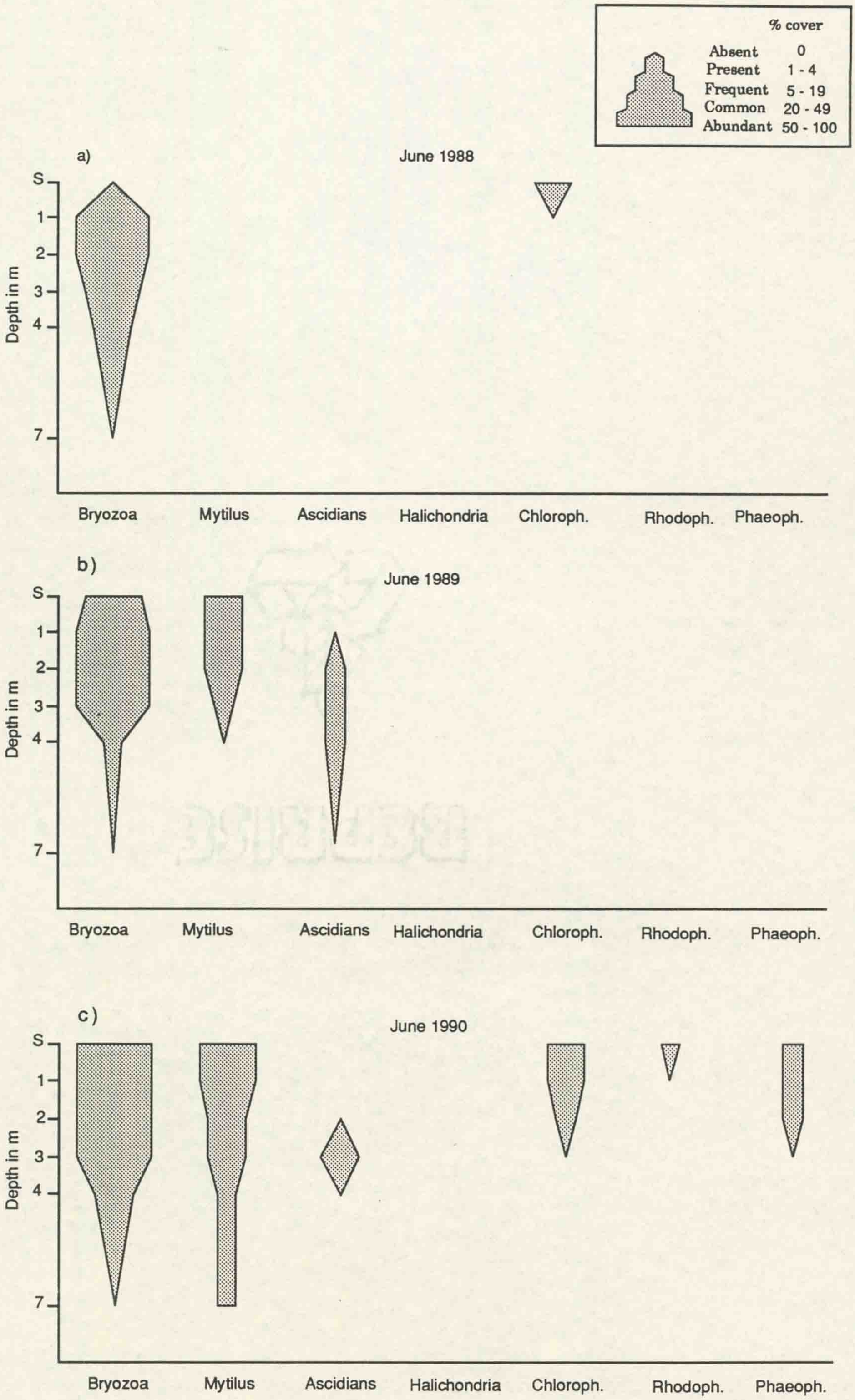


Fig 5.2 a to c Abundance of selected species with depth on the walls of the Graving Dock in June 1988, July 1989 and June 1990.

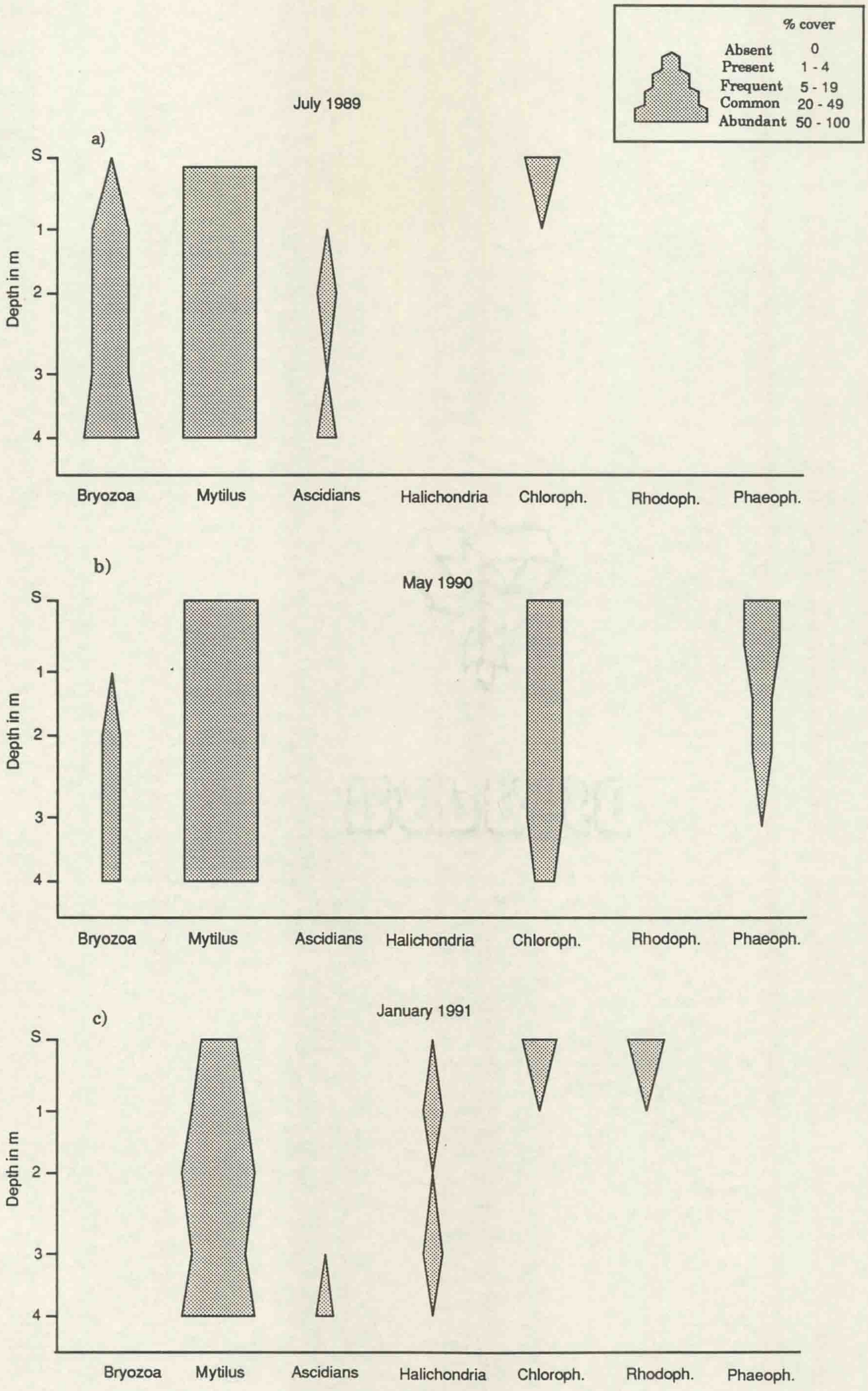


Fig 5.3 a to c Abundance of selected species with depth on the walls of the Albert Dock in July 1989, May 1990 and January 1991.

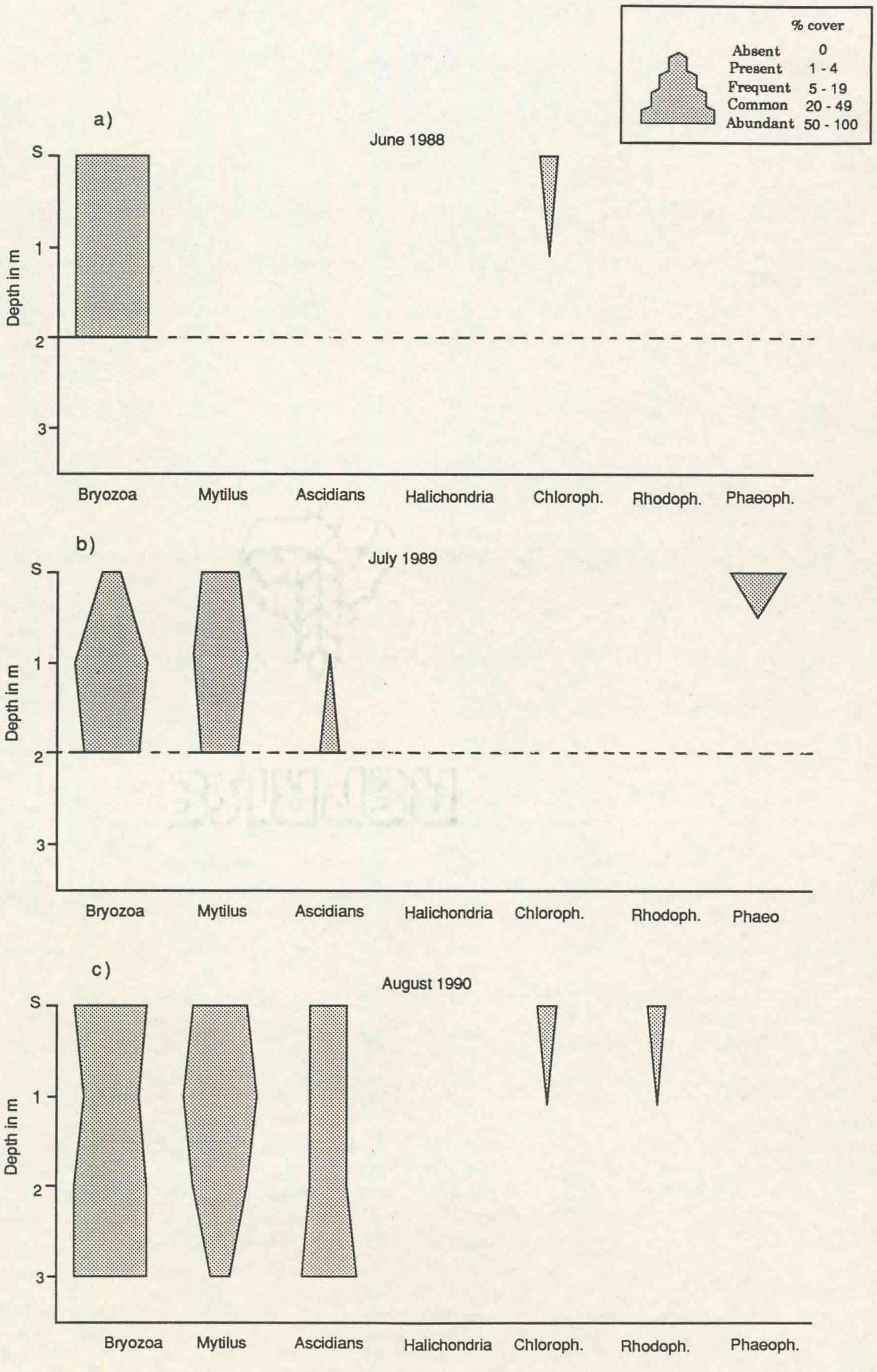


Fig 5.4 a to c Abundance of selected species with depth on the walls of the Queens Dock in June 1988, July 1989 and June 1990.
Dashed line indicates base of wall during period of low water

A single juvenile *Mytilus edulis* was found in each of the two docks. No fauna was present below 4 m depth in the Graving Dock. In June 1988 algae were restricted to a thin band of ephemeral green algae (mainly *Enteromorpha)* close to the surface.

In September 1988 a heavy settlement of mussels occurred in the South Docks. Examination of the dock walls in June/July 1989 showed this to be most dense in the Albert Dock and least in the Graving Dock (Figs 5.2 to 5.4). The ascidian *Molgula manhattensis* was also present by summer 1989 in small numbers. With time mussels gradually replaced bryozoans as the major species covering the wall in the Albert Dock, while in the Queens and Graving Docks bryozoan colonies survived between the more sparsely distributed mussels.

By January 1991 the sponge *Halichondria panicea* was present in small amounts in the Albert Dock at all depths (Fig 5.3c), but was not a notable component of wall cover in the Queens or Graving Docks. By January 1992 it had increased considerably in cover in the Albert Dock (S. Wilkinson pers. comm.).

On the earlier sampling occasions small green algae, mainly *Enteromorpha intestinalis* with some *Cladophora vagabunda* and colonial diatoms, were the only flora found. These plants only grew close to the surface. In later samples members of the Rhodophyceae (mainly *Ceramium* spp.) and Phaeophyceae (mainly *Punctaria latifolia*) became more common. With time algae penetrated to a greater depth on the dock walls in the Albert and Graving Docks (but not in Queens), reaching right to the bottom of the wall at 4 m depth on the Albert Dock (fig. 5.3b).

In summer 1990 the hydroid *Obelia dichotoma* was commonly found, often growing attached to mussels. Hydroid growth was particularly dense in Queens Dock.

Table 5.1a Combined species list macrofauna in South Docks. Field identifications, made whilst diving, are indicated * as they may be less reliable.

Fish and Benthic Fauna

Annelida - Polychaetes.
- *Arenicola marina*
- *Capitella capitata*
- *Nereis diversicolor* (Müller)
- *Nereis pelagica*
- *Nereis virens* (Sars)
- *Polydora ciliata*
- unid. Terebellidae

Bryozoa
- *Bugula simplex*
- *Conopeum seurati*
- *Conopeum reticulum*

Crustacea
- *Balanus improvisus*
- *Cancer pagurus*
- *Carcinus maenas*
- *Corophium insidiosum*
- *Crangon crangon*
- *Gammarus salinus*
- *Jassa marmorata*
- *Liocarcinus holcatus*
- *Microdeutopis gryllotalpa*
- *Praunus flexuosus*
- *Palaemonetes varians*
- *Porcellana platycheles*

Chordata

Ascidiacea
- *Ascidiella aspersa*
- *Botryllus schlosseri*
- *Ciona intestinalis*
- *Molgula manhattensis*
- *Styela clava*

Cnidaria
- *Aurelia aurita*
- *Metridium senile*
- *Obelia dichotoma*
- *Sagartia troglodytes*
- *Urticina felina*

Echinodermata
- *Asterias rubens*
- *Psammechinus miliaris* *

Mollusca
- *Angulus tenuis*
- *Facelina bostoniensis*
- *Littorina saxatilis* ag.
- *Mytilus edulis*

Porifera
- *Halichondria panicea*

Vertebrata
- *Anguilla anguilla*
- *Chelon labrosus* *
- *Ciliata mustela*
- *Clupea harengus*
- *Merlangus merlangus*
- *Gadus morhua*
- *Gasterosteus aculeatus*
- *Limanda limanda*
- *Taurulus bubalis* *
- *Platichthys flesus*
- *Pleuronectes platessa*
- *Pomatoschistus microps*
- *Solea solea*
- *Sprattus sprattus*
- *Sygnathus rostellatus* *
- *Trisopterus luscus*

Table 5.1b Combined species list, macroflora in South Docks

Rhodophyceae
- *Antithamnion plumula*
- *Audouinella secundata*
- *Ceramium deslongchampii*
- *Ceramium rubrum*
- *Erythrotridia carnea*
- *Polysiphonia urceolata*
- *Porphyra purpurea*

Phaeophyceae
- *Ectocarpus siliculosus*
- *Fucus vesiculosus*
- *Giffordia ovata*
- *Pilayella littoralis*
- *Punctaria latifolia*
- *Sorocarpus micromorus* (det. R. Fletcher)
- *Stictyosiphon soriferus*

Chlorophyceae
- *Bryopsis hypnoides*
- *Cladophora vagabunda*
- *Enteromorpha compressa*
- *Enteromorpha intestinalis*
- *Enteromorpha linza Var. lanceolata* (det.P. Kornmann)
- *Monostroma grevillei*
- *Ulothrix pseudoflacca*
- *Ulva lactuca*

Chrysophyceae
- *Vaucheria litorea.* (det. T. Christensen)

A variety of small animals were found in the mussel/bryozoan matrix (see Appendix Tables XV to XVII). The barnacle *Balanus improvisus* and the colonial sea squirt, *Botryllus schlosseri* were particularly common attached to mussel shells. Several species of *Nereis* were found and *Polydora ciliata* was often abundant living in tubes made from detrital material.

In 1989 and 1990 amphipods were often abundant, particularly in summer months. *Corophium insidiosum* and *Microdeutopis gryllotalpa* were the dominant species. *Jassa marmorata*, a common species in the Queens Dock was only occasionally found in the Albert Dock. The highest densities of amphipods were normally found in Queens Dock.

Several less common large benthic species were recorded during timed searches (Appendix Table XVIII) and other diving operations (see species list, Table 5.1). These include the ascidian *Styela clava* which was observed for the first time in summer 1990. This species, which is native to Kórea and Japan, was introduced relatively recently to the British Isles and was described by Millar (1970) as being found only in Southern Britain. This species was also found in Princes Dock where it was relatively abundant. Several species of anemone were present in the docks: *Metridium senile*, both large and dwarf forms were commonly encountered; additionally *Urticina felina* was occasionally found especially in the shade provided by pontoons; *Sagartia troglodytes* was frequently found protruding through the sediment, but attached to the dock wall, at the wall / sediment interface.

5.3.2 Wall Fauna Of Other Docks

The dominance of mussels in the South Docks is unusual when compared to other docks on the lower Mersey Estuary (Table 5.2). The species lists in table 5.2 are based on very small samples of the dock walls and do not give an exhaustive species list for each dock, however they are indicative of the main species present on the dock walls. The fauna of most other docks studied in summer 1990 was dominated by the sea squirts *Ascidiella aspersa* and *Ciona*

Table 5.2 Salinities and epifaunal species present in wall scrapes, from seven docks on the lower Mersey Estuary, summer1990. Modified from Wilkinson, Allen and Hawkins 1990.

	Bidston	Hornby	Sandon	Stanley 1/2 Tide	Princes	Albert	Queens
Salinity o/oo	29.0	29.8	28.0	18.5	28.0	26.0	26.0
Mytilus edulis	-	-	-	-	P	A	A
Molgula manhattensis	A	-	-	-	P	A	P
Ciona intestinalis	P	-	-	P	P	-	-
Ascidiella aspersa	P	A	P	P	P	-	-
Botryllus schlosseri	√	√	√	√	√	-	-
Buglula simplex	√	√	√	√	√	-	-
Conopeum reticulum	√	-	√	√	√	√	√
Cryptosula pallasiana	-	-	-	-	√	-	-
Balanus improvisus	√	√	√	√	-	-	√
Obelia dichotoma	√	√	√	√	√	-	-
Carcinus maenas	√	√	√	-	√	-	-
Nereis diversicolor	-	-	-	-	-	√	√
Polydora ciliata	√	√	√	√	√	-	√
Corophium insidiosum	√	√	√	√	√	√	√
Gammarus salinus	√	-	-	-	-	√	-
Jassa marmorata	√	√	√	√	-	-	√
Microdeutopus gryllotalpa	√	√	√	√	√	√	√
Metridium senile	-	√	-	-	-	-	-

Key :-
- Absent ; P Present ; A Abundant

Classification categories of abundance for wall scrapes

Species	Total no. in 0.08m2 (= area of 5 repl. scrapes) Present (P)	Abundant (A)
Solitary tunicates, *Mytilus*	1-5	5+

All other groups recorded simply as present (√) or absent (-).

Table 5.3 Sediment fauna of Graving (G), Albert (A), Queens (Q) and Brunswick Docks (B), collected in grab samples, Aug. 1989 to Nov 1990. Numbers per square metre, calculated from mean of three replicate grabs.

Date	August 1989				May 1990			November 1990		
Dock	G	A	Q	B	G	A	Q	G	A	Q
Arenicola marina									4	
Capitella capitata		35	58	266					417	12
Nereis pelagica							3			4
Nereis virens							23			12
Polydora spp.						138	38		193	
Ampharetidae (Unid.)									8	
Oligocheate (Unid.)		6	12	12						
Corophium insidiosum									8	
Microdeutopis gryllotalpa				3						
Molgula manhattensis						8	4		8	170
Angulus tenuis									4	
TOTALS	0	41	70	281	0	146	68	0	642	198

intestinalis, with mussels found only in Princes Dock. *Ascidiella* was very rarely found in the South Docks.

5.3.3 Sediment Benthos

No live fauna was found in the Graving Dock on any sampling occasion (Table 5.3). The sediment collected in grabs was of fine, unconsolidated material, completely black and anoxic and smelling strongly of hydrogen sulphide.

In the Albert and Queens Docks small numbers of animals were found in all grabs taken (Table 5.3). Sediments in these Docks were also very fine-grained and had a high proportion of anoxic material, but they were better consolidated and had a grey oxic layer of varying thickness. In the earliest samples taken in August 1989, the polychaete *Capitella capitata* was the most common species found, with occasional oligochaetes and amphipods. In May 1990 no *Capitella* was found and the spionid polychaete *Poydora ciliata* was now the numerically dominant species, with occasional individuals of the tunicate *Molgula manhattensis*. In Queens Dock alone, *Neries* spp. were frequently found.

In November 1990 the densities of animals found in both the Albert and Queens Docks had increased and in the Albert Dock a greater number of species were present. *Capitella capitata* and *Polydora ciliata* were found in both docks, with *Capitella* more frequent in Albert and *Polydora* more frequent in Queens. *Nereis* spp. was again present in Queens but not in Albert Dock. In November 1990 *Molgula manhattensis* was present in quite high numbers in Queens Dock but was still in low densities in Albert Dock. Three previously unrecorded species *Arenicola marina*, the bivalve *Angulus tenuis* and an unidentified terebellid worm were collected in grabs from the Albert Dock.

No macroalgae were collected in grab samples. However observations during diving operations showed that a variety of algae were present attached to debris on the dock bottom.

Ulva lactuca was often found growing attached to mussel clumps that had fallen from the wall. In Salthouse Dock, (3.5m deep, adjacent to Albert Dock) a thick mat of algae, mainly *Vaucharia litorea*, covered large areas of the sediments from early summer 1989 onwards, but this was not observed in other docks.

5.3.4 Fish, Nekton and Mobile Benthos

Species of fish, nekton and mobile benthos collected in the South Docks are included in table 5.1a. A total of 16 species of fish were found. Nettings carried out by, or in conjunction with, Manchester University showed a dominance of whiting (age groups 1+ and 2+) in summer samples in both 1988 and 1989, with large numbers of sprats caught in nets laid out in December1988. Plaice and flounder (age groups 1 - 3+) were also commonly caught species (Mincher 1988, Heaps 1989, Lonsdale 1990). The sand goby *Pomatoschistus microps* was the fish most commonly encountered during diving surveys. This species formed burrows in the sediments on the dock bottom.

Of the invertebrate nekton, the common jellyfish, *(Aurelia aurita)* was the most notable species in summer. Large individuals often occured in densities of >5 per cubic metre locally. Shoals of the mysid *Praunus flexuosus* were frequently found close to the dock walls or in shallower areas.

Common mobile benthic fauna included *Carcinus maenas* and *Crangon crangon*. *Cancer pagurus* was very rarely found, possibly due to the scarcity of suitable habitat. *Asterias rubens* was only found on one occasion in the Graving Dock in spring 1989, living on introduced mussels. *Asterias* was observed in other docks in the Mersey Estuary during the survey in summer 1990, however, it did not appear to thrive in the South Docks after its inadvertant introduction.

Table 5.4 Mean length, mean dry weight and density of mussels on the walls of the Albert, Graving and Queens Docks, 1989 to 1991. Mean dry weights calculated from length/dry weight regression equations. Mussel densities calculated from the means of three 0.25 x 0.25 m quadrats.

Date	Dock	Mean Length (mm) ± SD (sample no.)	Mean Dry Weight Of Soft Parts (g) ±SD	Number Of Mussels m^{-2} On Dock Wall At 1m Depth
28-7-89	Albert	25.6 ± 6.27 (200)	0.08 ± 0.038	4576
29-8-89	Graving	30.0 ± 19.3 (24)		112
4-8-89	Queens	40.2 ± 5.8 (61)	0.44 (pooled samp.)	976
27-11-89	Albert	30.6 ± 9.38 (200)		4992
30-11-89	Graving	47.2 ± 11.05 (48)		55
7-11-89	Queens	40.2 ± 9.98 (157)		251
23-5-90	Albert	33.0 ± 7.76 (251)	0.131 ± 0.112	5456
1-6-90	Graving	39.9 ± 16.83 (98)	0.522 ± 0.475	69
24-5-90	Queens	50.2 ± 8.63 (216)	0.839 ± 0.353	1157
15-8-90	Albert	41.3 ± 7.41 (201)	0.281 ± 0.198	1995
22-8-90	Graving	41.2 ± 13.78 (150)	0.424 ± 0.299	59
24-8-90	Queens	50.1 ± 14.57 (200)	1.127 ± 0.648	677
8-1-91	Albert	41.1 ± 7.95 (205)	0.333 ± 0.226	

5.3.5 Natural *Mytilus edulis* Populations

The initial settlement of the mussel *Mytilus edulis* occurred in September 1988. These mussels were not studied in detail until summer 1989. At this time only one cohort of mussels was present in the docks (Figs 5.6a and 5.7a). Initial settlement was most dense in the Albert Dock and mussel density remained highest in the Albert Dock throughout the period of study (Table 5.4). By summer 1989, after just less than one years growth, mussels had reached an average size of 25.6 mm in Albert Dock and 40.2 mm in Queens Dock. In all samples taken the mean lengthof mussels in Albert was less than in Queens Dock with the Graving Dock having an intermediate size (Table 5.4). The lengths acheived by the initially settling (1988) cohort of mussels, after 20 months, as estimated from Figs 5.5 to 5.7 is compared in table 5.5 to growth of subtidal mussels from other temperate regions.

The mean tissue weight to shell weight ratio of a mussel population is dependent on environmental conditions such as exposure to waves and air or food supply (Seed 1973). In the South Docks exposure varies little from dock to dock, so this ratio may be used to indicate differences in condition of musssels between docks. The mean ratio of dry weight of soft parts to shell weight was 0.1024 (S.D. 0.052) in Albert Dock compared to 0.1670 (S.D. 0.061) in Queens Dock for May 1990 samples. The difference between these two means was tested by means of a Students *t*-test and was found to be significant at the 0.1% level. Hence mussels from Queens Dock have significantly more meat for a given shell weight.

The occurrence of spat settlement can be clearly seen in the length-frequency histograms (Figs 5.5 to 5.7). After the first settlement (September 1988) spat also settled in all three docks in late summer 1989. Samples taken in summer 1990 show that spat settlement in this year occurred between May and late August in Queens and Graving Docks, but no recruitment was seen in Albert Dock. The main autumn 1990 settlement season appeared to fail altogether in Albert Dock, no juvenile mussels were present even by January 1991 (Fig

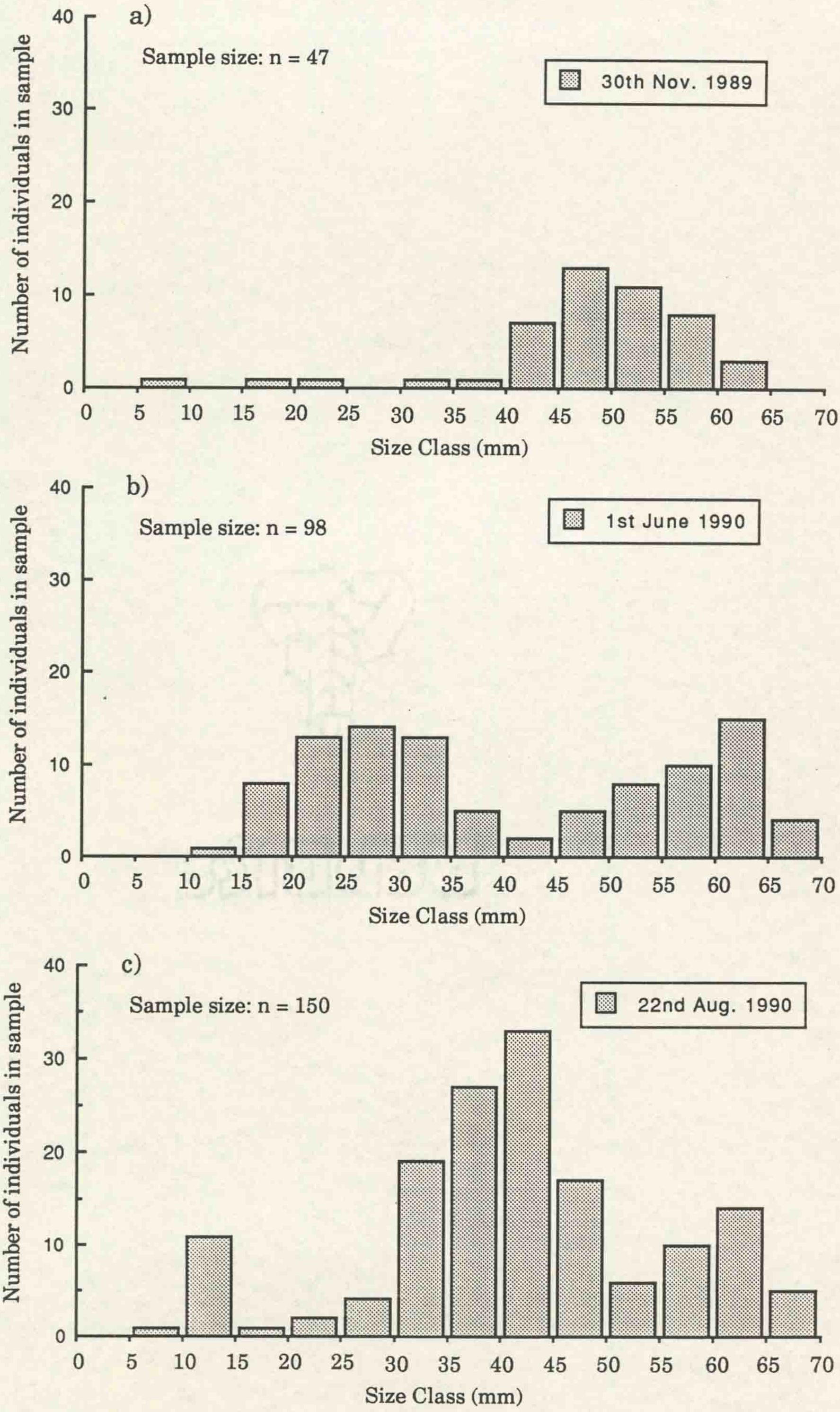


Fig. 5.5 Length / frequency distribution of mussels in the Graving Dock (surf. to 4m depth). a) November 1989, b) June 1990 and c) Aug 1990.

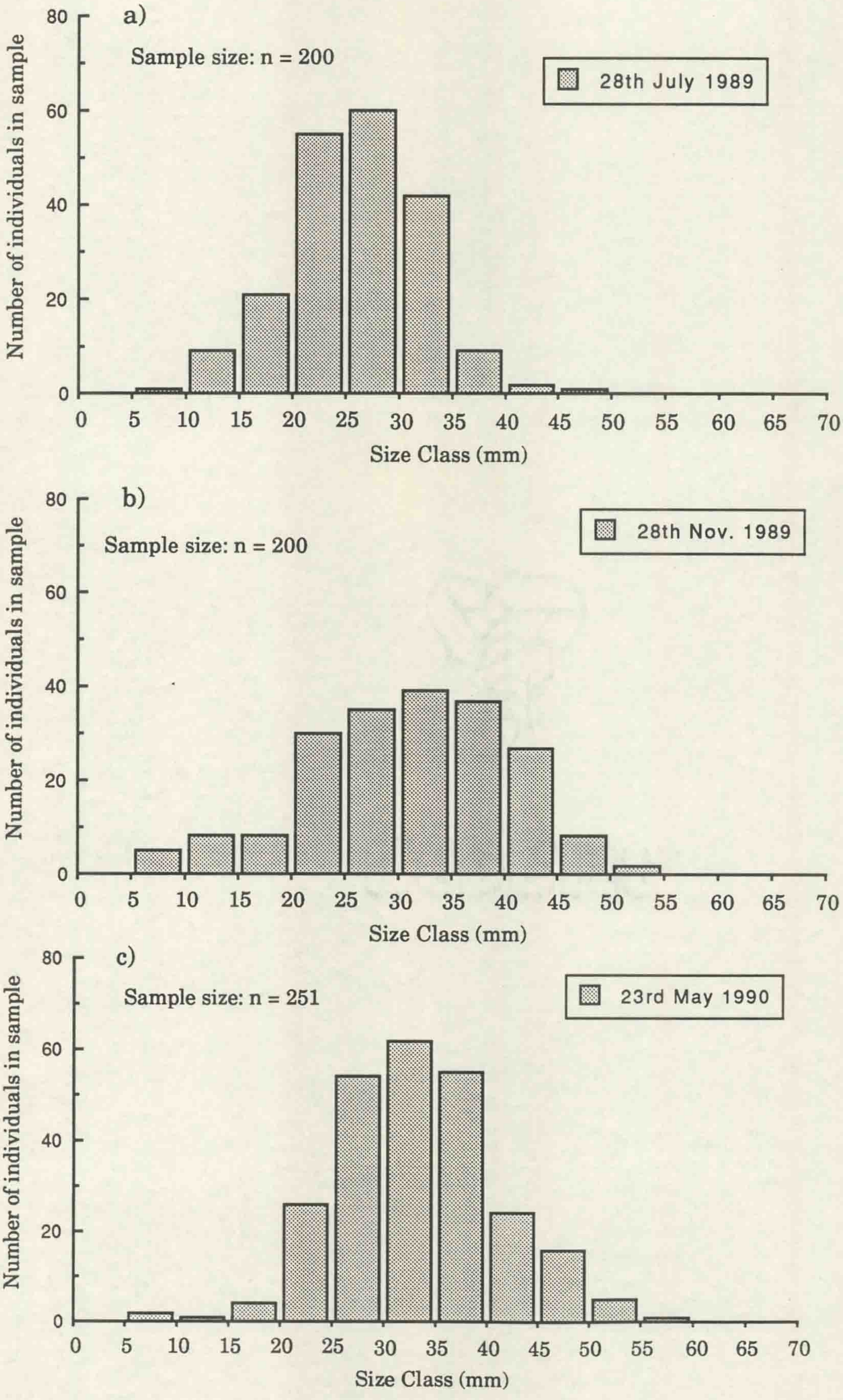


Fig 5.6 Length frequency distribution of mussels in the Albert Dock (from 1m depth). a) July 1989, b) Nov. 1989, c) May 1990. Note change in scale due to differences in sample size.

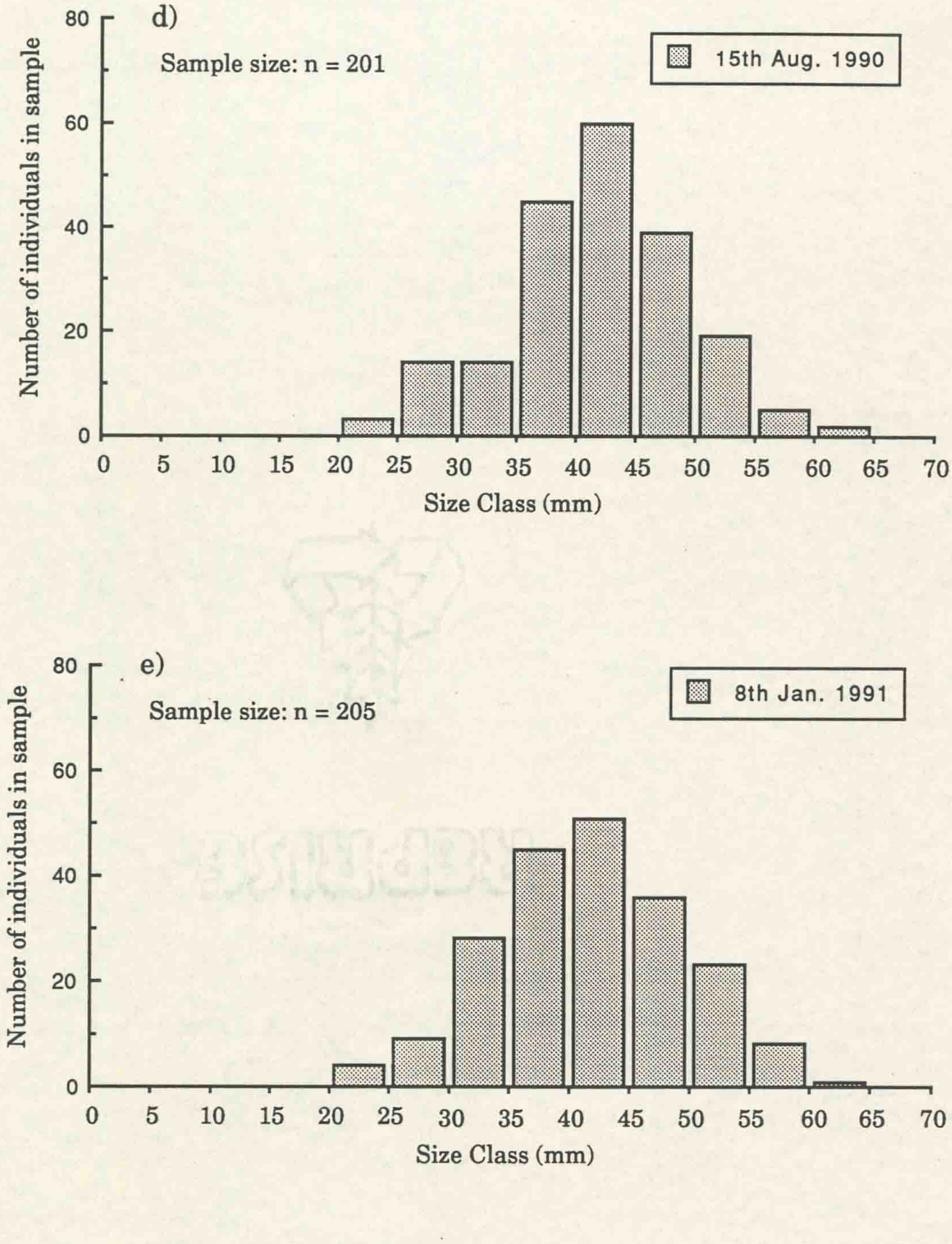


Fig 5.6 Length / frequency distribution of mussels in the AlbertDock (from 1m depth). d) 15th Aug 1990, e) 8th Jan 1991.

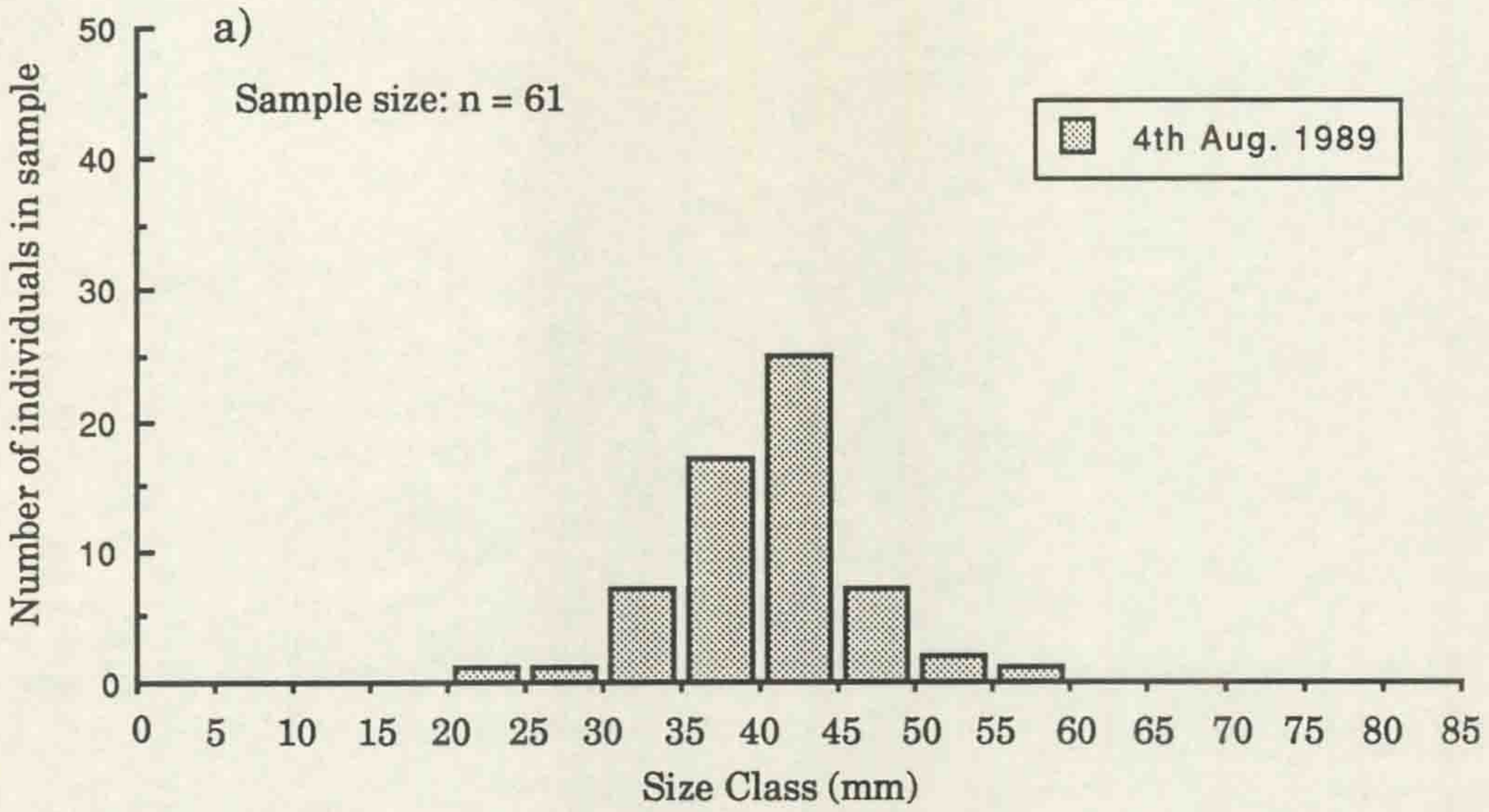
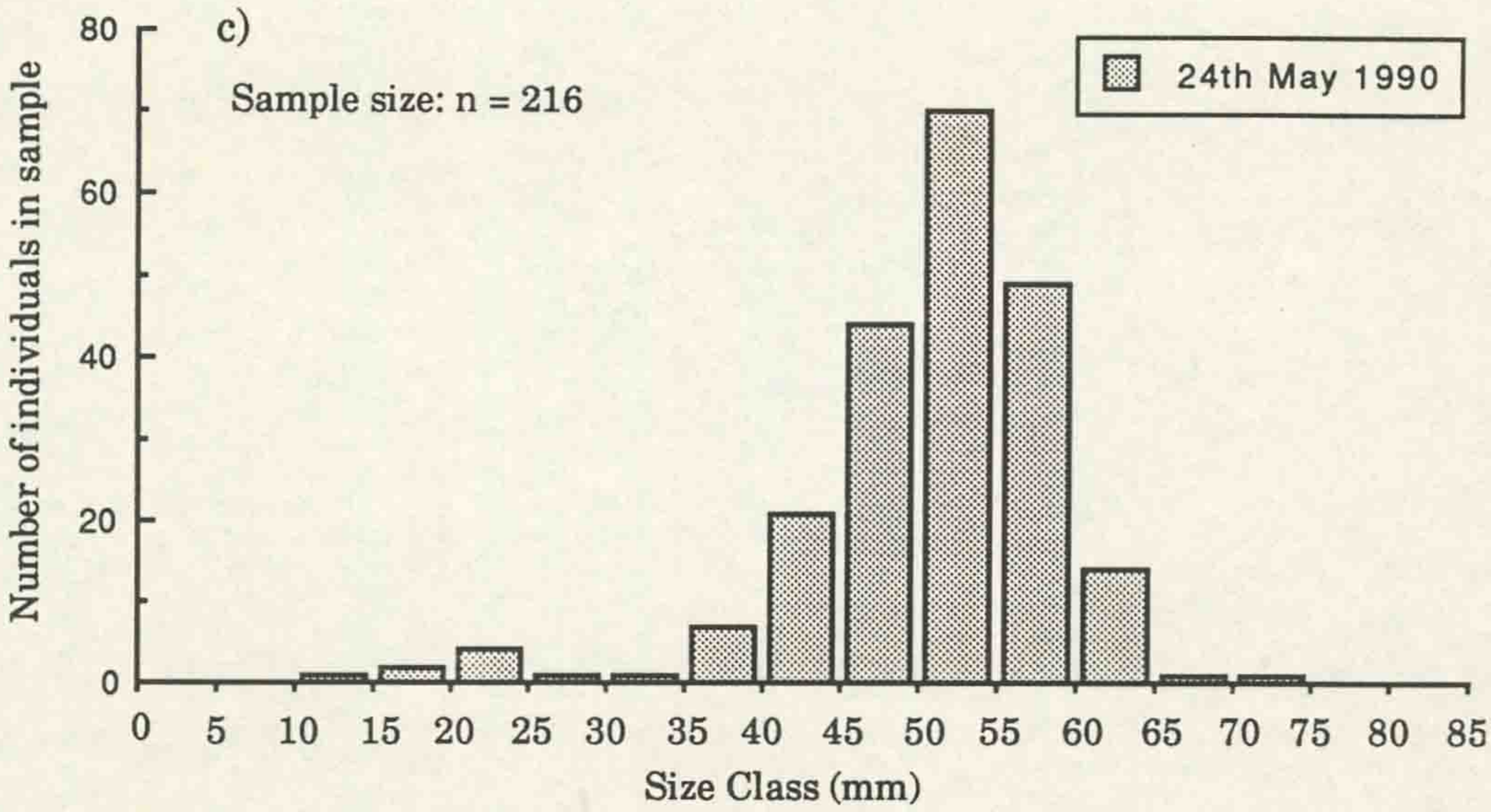


Fig 5.7 Length / frequency distribution of mussels in the Queens Dock (from 1m depth). a) Aug 1989, b) Dec 1989. Note change in scale due to differences in sample size.



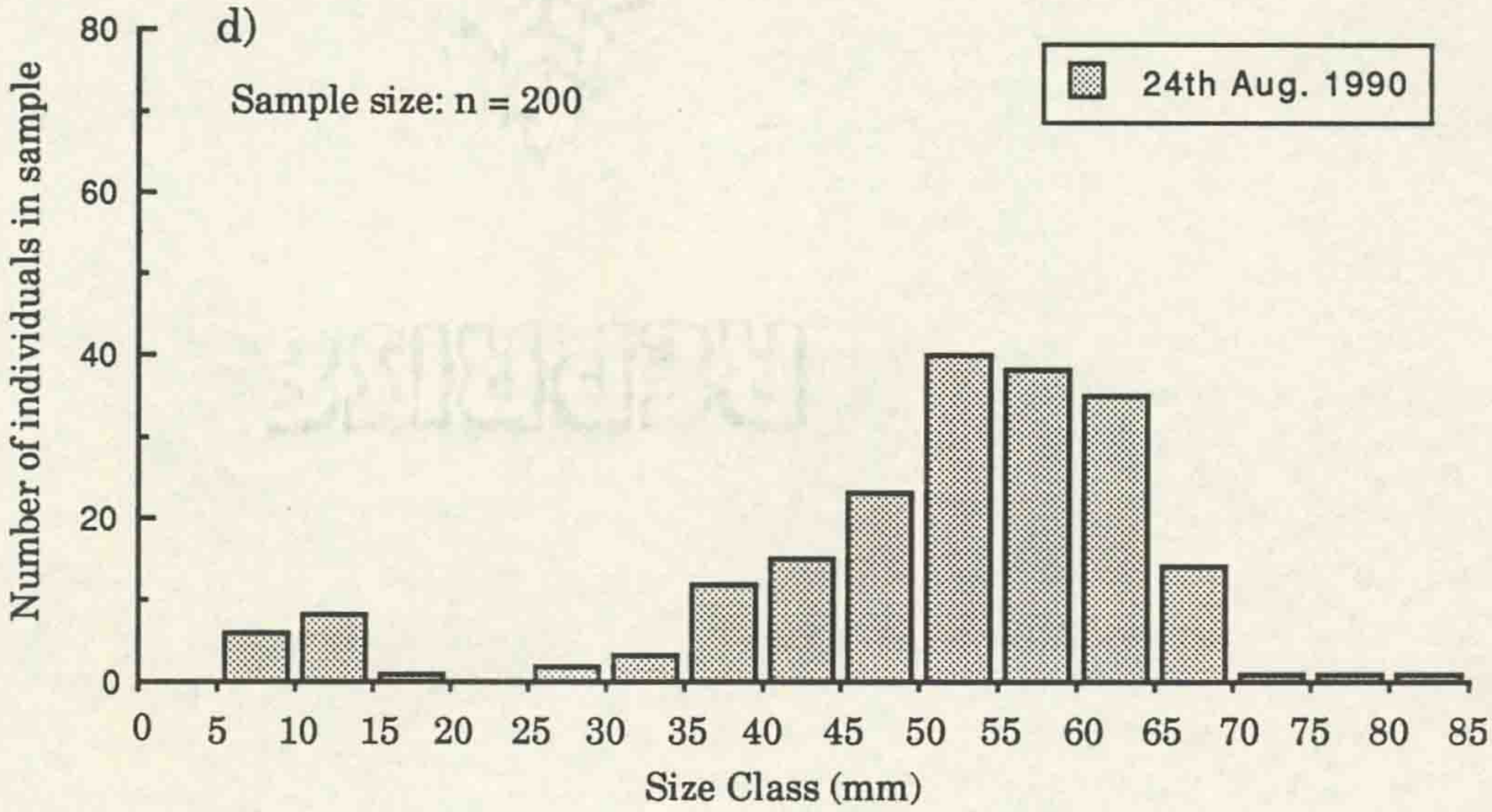


Fig 5.7 Length / frequency distribution of mussels in the Queens Dock (from 1m depth). c) May 1990, d) Aug 1990. Note change in scale due to differences in sample size.

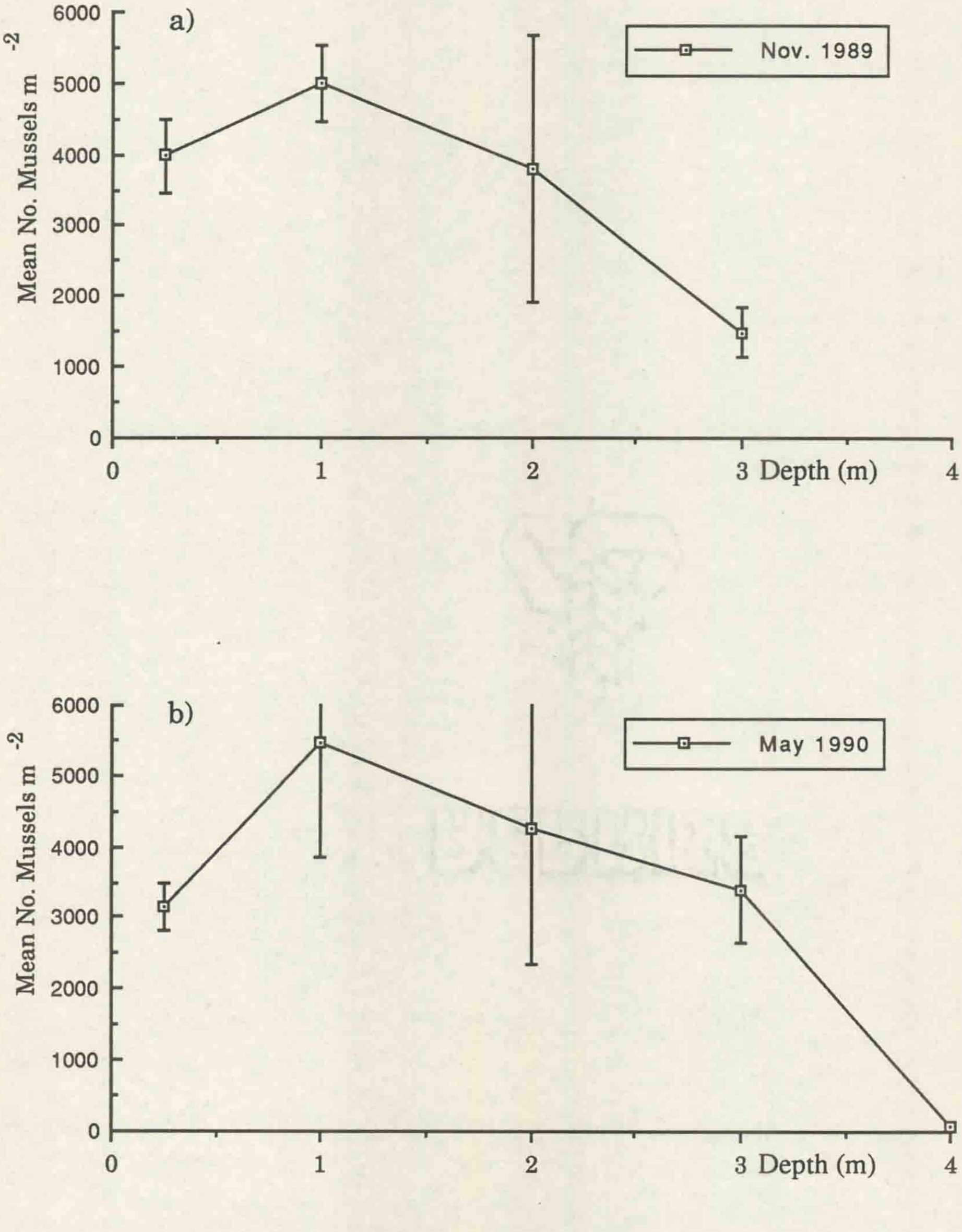


Fig. 5.8 Density of mussels on the Albert Dock Wall with depth. ± S.D. a) November 1989, b) May 1990. Mean densities calculated from 3 replicate 0.25 x 0.25 cm wall scrapes.

5.6e). This may be due to failure of recruitment rather than larval supply because empty juvenile shells were noted in samples from the Albert Dock in August 1990.

The separate cohorts for all three years can be most clearly seen in the Graving Dock samples (Fig 5.5 a to c) and are most difficult to distinguish in the Albert Dock where growth seems to be limited. The presence of mussels of the 10 - 15 mm size class in all docks in May 1990 suggests that a small amount of spat settlement may occur throughout the year, probably by secondary settlement, although the main release is in late summer.

Mussel cover is not constant with depth, tending to be most dense at around 1m and least dense close to the dock bottom (Fig 5.8).

5.4 DISCUSSION

At the start of the project only one year of funding was envisaged and consequently studies of the benthos were of low priority. Much of the sampling was incidental to other tasks. Hence the level of replication of samples was low and statistical comparisons between sites, or over time, cannot be made. In retrospect it would have been valuable to have carried out more intensive surveys at earlier dates as some important stages in succession were missed. The unforeseen rapid changes in communities that took place so shortly after the beginning of the sampling programme prompted a more detailed study of the benthos than was originally planned. Despite the limitations of the work the overall pattern of colonisation can still be described.

The first observations on wall communities carried out in Queens and Graving Docks took place approximately four years after water had been returned to these docks (see chapter 2). The somewhat irregular studies carried out after this time showed a general progression

from communities dominated by bryozoans to mussels and then to more mixed communities, with macroalgae, ascidians and finally sponges becoming more common. *Mytilus* almost completely replaced bryozoa in the Albert Dock, but was less dominant in the Queens and Graving Docks.

Scheer (1945), in a study of the fouling communities of the undersides of floats, found successional patterns and species assemblages which were similar in many ways to the South Docks. A progression from bacteria and protozoa to diatoms, hydroids, bryozoans, ascidians (*Ciona* and *Styela*) and finally to *Mytilus* occurred, with seasonal variations due to mass settlement of barnacle or ascidian larvae and subsequent mortality. As in the Albert Dock sponges were often associated with a community in which ascidians were common. Similar patterns to those seen in the South Docks include studies using settlement plates. For example, Chalmer (1982) described a progression from *Balanus* and *Spirorbis* to bryozoan and *Mytilus* dominated fauna. Dean and Hurd (1980) also found that barnacles and serpulids settled on bare surfaces and that the presence of hydroids enhanced settlement of ascidians, while the ascidian/hydroid assemblage enhanced settlement of mussels. Succession on a short time scale, which included a barnacle, hydroid and/or ascidian dominated phase before mussel settlement as described in the studies above, may have occurred in the South Docks, but has been missed as no sampling was done before this study commenced in 1988. Some evidence for this is that empty *Balanus improvisus* shells were present underneath other encrusting fauna from the first time of sampling, it is unlikely that these persisted from before redevelopment as they extended to depths which would have been below the level of mud when the Docks were disused. Also, frequent examination of mesh ropes placed in the Albert Dock in July 1989 to collect mussel spat showed a progression from hydroids to ascidians (*Molgula manhattensis*) to *Mytilus* by late September of the same year with almost complete cover of ropes by each species in every case. Had such a succession occurred on the South Dock walls over the same period in 1988 this would not have been observed by sampling.

The temporal changes and spatial differences seen in wall communities are likely to be due to a combination of factors including succession by classical facilitation, tolerance and inhibition models, variations in propagule supply and competition. For example, settlement of mussels and ascidians is enhanced by the initial presence of hydroid or filamentous algal cover (Bayne 1964, Chalmer 1982, Stocker & Bergquest 1987, King *et al* 1990). *Mytilus* has been found to be the climax species in several studies on succession with such domination achieved by an ability to settle in established communities and outlive other species (Dean & Hurd 1980, Dean 1981, Chalmer 1982). Hence, it is likely that initial colonisation of the South Docks by mussels was brought about by facilitation of primary settlement by earlier colonisers. Subsequently the tolerance of *Mytilus* for co-existing species, and also probably the inhibition of settlement of some species, would have ensured its continued survival. The longevity of *Mytilus* compared to other competing species may have ensured its eventual dominance. In the South Docks larval supply is controlled by both seasonal variation and timing of water intake. Introductions of new species to the dock take place if larvae are present in water used for topping up. As topping up is only carried out once every two weeks, at a limited state of the tide, species whose larval release does not coincide with this timing and appropriate estuarine currents may be precluded from the South Docks.

Conspecific competition for space and/or food by *Mytilus* is the most likely explanation for the increased patchiness and diversity seen in the Albert Dock towards the end of the study. In this way the mussel population in the South Docks may become self regulating with time. Mussels became so dense in this Dock that they began to grow over the top of each other and eventually began to fall away from the wall in clumps, thus opening up space for colonisation by other species. This led to a more diverse community with a patchwork of other species exploiting the available space. In this way competition on the vertical walls has a comparable effect to that associated with physical disturbance or predation. The increased densities of *Ciona intestinalis* seen in the Albert Dock in later surveys may also be due to some competitive advantage in the lower food concentrations, for example better filtering ability or lower energy requirements. *Ciona* also became more prevalent in Sandon Dock when food

supply decreased (Naylor 1983, Cunningham *et al* 1984). The increased depth penetration of algae seen with time is related to the increased water clarity over the same period.

Large seasonal fluctuations in numbers of some species were also evident, particularly in the case of ephemeral algae, amphipods and *Molgula*. This may be due to a short life span in some species. In particular many algae are annual in nature and die off completely in the winter. Seasonal changes in environmental conditions and food supply will effect the survival and production rates of longer-lived species. Seasonal larval settlement, with subsequent failure of recruitment, elevated the populations of some species (e.g. *Molgula*) for periods.

The differences between the composition of wall communities seen between docks on the Mersey Estuary is somewhat surprising given the similarity of the systems. *Mytilus* was more frequent in disused docks such as Sandon, Princes and the South Docks, while *Ascidiella aspersa* was more common in the operational docks. Such differences may be a function of larval supply, however differences in water quality parameters, such as suspended solids, may have a direct or indirect effect on recruitment.

Spatial differences in wall communities throughout the South Docks tend to be in terms of relative densities of particular species. Such variation may be due to small scale patterns of water movements and larval supply or to the different length of time since water has been returned to each dock (see chapter 2). In the Graving Dock the use of hydrogen peroxide to combat a problem of foul smelling water in April 1988 caused a fish kill (M.D.C. pers comm.) and is likely to have affected the biota on the walls. The use of an artificial mixer in this dock prevented such bad smells and also allowed fauna to survive at greater depths by increasing the oxygenated zone.

In the South Docks poorly oxygenated waters and sediments will restrict the diversity and

distribution of fauna. The complete absence of fauna in the sediments of the Graving Dock is not surprising given the persistent low oxygen levels in bottom waters, even with aeration. Dead or dying polychaetes and fish in respiratory distress were observed in the Albert Dock during a period of anoxic conditions in summer 1989. The appearance of large polychaetes and bivalves in sediment samples from this dock only occurred in autumn 1990 after a summer with no prolonged anoxic periods. Oxygen concentrations may have small scale effects on distribution, for example, in Queens Dock *Molgula manhattensis* living on the dock bottom tends to inhabit raised portions of sediment . Spatial variations in sediment fauna within the South Docks may also be due to differences in sediment type (James & Gibson 1980) or larval supply.

The appearance of mats of filamentous algae over sediments in Salthouse Dock may have implications for water quality. Such mats, when dying back in winter, could cause localised oxygen depletion, or float to the surface to form scums as gases are released. No floating mats were observed but this is a potential problem.

Chance events such as extreme weather conditions, low oxygen levels, pollution incidents and large fluctuations in larval supply, which could potentially have a major impact on the ecosystem of the South Docks, make the concept of a stable climax community questionable. Without such catastrophic events, however, it is likely that *Mytilus* will continue to be an important component of the communities of the dock walls, given its longevity and apparently plentiful larval supply. It is likely however that patchiness and diversity of the wall communities will continue to increase with time due to competition and predation effects as described previously. The mussel spat that settled in the Albert Dock in summer 1990 was not able to survive. This was possibly due to a shortage of food or space caused by the dense population of mature mussels. The benthic plant and animal communities of the South Docks continue to develop with new species still appearing (I Wallace, pers. comm.). Such colonisation is restricted to species which are tolerant of the extremes of water temperature and reduced salinity which characterise the dock environment. Low water movement may

SITE	AGE (months)	MODAL LENGTH (mm)
Graving Dock	20	60 to 65
Albert Dock	20	30 to 35
Queens Dock	20	50 to 55
Sandon Dock *	24	50 to 55
Sweden ◊	17 - 18	50 (mean)
Scotland /Wales ◊	16 - 18	50 - 60

Table 5.5 Growth of mussels in the South Docks compared to published figures for subtidally grown mussels in other areas.

* From Hawkins *et al* a in press.

◊ From Loo & Rosenberg 1983.

cause problems, leading to mortality in some cases, for example by facilitating the growth of epiphytes on macroalgae or inducing bacterial growth on sponges (Hummel *et al* 1988). The lack of three dimensional variety of structure afforded by the featureless dock walls may exclude crevice dwelling species. It is obviously difficult for species with no free living dispersal phase, such as *Nucella lapillus*, to penetrate the dock system, although this may happen eventually by rafting on debris.

Two species are worthy of note as being possibly unusual for the area. The amphipod *Microdeutopis gryllotalpa* is recorded only from South West Britain in Lincoln (1979) and the algae *Sorocarpus micromorus* is very rare in the British Isles. Three species found in the South Docks were listed amongst specialist lagoonal species by Barnes (1988), these are *Conopeum seurati*, *Corophium insidiosum* and *Palaemonetes varians*. In the Mersey Estuary *Corophium volutator* is very common, but *Corophium insidiosum* is rarely found (J. McGill pers. comm.). Interestingly, no *C. volutator* was found in the South Docks but dense populations of *C. insidiosum* were present. The South Docks could possibly provide a suitable habitat for other lagoonal species.

The sixteen species of fish found in the South Docks have all been recorded recently from the Mersey Estuary (Wilson *et al* 1988, Lonsdale 1990) and are all typical estuarine or coastal species. Lonsdale (1990) suggested that seasonal variations in numbers of sprat caught in the South Docks indicated a migration typical of estuarine waters. It is very likely that large numbers of sprat enter the the docks when abundant in the estuary in winter. However, their subsequent migration from the docks is difficult to explain as direct water movement from the river is almost exclusively inwards, apart from small amounts lost during the locking out of boats. Predation by birds and larger fish may therefore play the major role in the decline of sprat populations over summer months.

The growth rates for Queens and Graving Dock mussels are comparable to other published figures (e.g. Hawkins *et al* in press a, Loo & Rosenberg 1983, see Table 5.5). The growth rate

in Albert Dock is much lower, possibly due to competition for either space or food on the more densely populated walls. In a study of the oyster *Crassotrea virginica* Zajac *et al* (1989) found that both food supply and density of competitors could independently affect the growth and survival of newly settled individuals. With the complete cover of *Mytilus* on the walls of the Albert Dock it is likely that competition for space was reducing growth, however the significantly lower meat to shell weight ratios found in this dock suggests that food supply was also a limiting factor.

In summary, a community has developed in the South Docks that is relatively diverse for a stretch of water positioned on a polluted estuary. This community is still changing, in particular, the diversity of the macroalgal, filter feeder and sediment benthos assemblages seems to be increasing and a more patchy distribution of macrofauna appears to be developing on some of the dock walls. This work presents a preliminary study of colonisation and community development in a polyhaline redeveloped dock. A more detailed study is required to determine the precise pattern of succession, in particular that of the initial stages, which were missed by this study.

SECTION 2
EXPERIMENTAL ECOLOGY

CHAPTER 6
MANAGEMENT STRATEGIES

6.1. INTRODUCTION

The development of the South Docks for cultural activities, recreation, housing, tourism and prestige office space relies heavily on its waterside location. Good water quality is essential for its economic success. Two main water quality problems were identified in chapters 2 and 3. Firstly, phytoplankton blooms cause discoloration of the water, and potentially toxic species are sometimes present. Secondly, periods of low oxygen concentrations in bottom waters occur in summer, which have been responsible for the release of foul smells and mortality of fauna. Such problems are linked to the source of the water in the South Docks. The waters of the Mersey Estuary are nutrient rich, bear a heavy load of sediment and are contaminated by raw sewage discharge (NRA data). Clearly the development agency (MDC) needs to solve the aesthetic and public health problems in order to maximize the appeal of the waterside location and associated economic success.

In freshwater lakes and reservoirs the occurrence of low oxygen levels during periods of thermal stratification is frequently encountered. In lakes where this presents a major problem water mixers of various designs have been used successfully to improve oxygen concentrations (Fast 1973, Bailey-Watts *et al* 1987). Water mixers may work on a hydraulic or pneumatic system (see reviews in Tolland 1977, Pastorok 1981). Hydraulic systems involve the direct transfer of water by water jets or paddles while pneumatic systems release air bubbles into the hypolimnion which entrain water as they rise to the surface, thus setting up a circulation pattern. Both systems work mainly by atmospheric aeration of water brought up from the hypolimnion. Pneumatic systems are generally cheaper to run and easier to operate (Pastorok 1981); such devices tested in lakes and reservoirs include perforated pipes, diffuser domes and bubble guns. One commercial air-driven mixer, the 'Helixor' passes air bubbles initially through a moulded helical insert within a tube. This gives the bubble plume a swirling component which increases oxygen transfer and entrainment of water (Henderson-Sellers, 1984). Helical type mixers have been used successfully in several docks (Russell *et al* 1983, Radway *et al* 1988).

There is no doubt that artificial mixers can improve oxygen concentrations, however, the direct effects of increased mixing on phytoplankton populations are not clear. Mixing has been reported to reduce phytoplankton populations in two ways. In deeper lakes the phytoplankton may be transported below the photic zone where light will limit growth (Pastorok *et al* 1980). Additionally the increased dissolved oxygen content of the water may result in reduced phosphate release from the sediments, hence increasing the possibility of nutrient limitation of growth (Pastorok *et al* 1981). Some studies have reported a decrease in phytoplankton biomass with mixing (e.g. Reynolds *et al* 1984, Bailey-Watts *et al* 1987), while others report increases (e.g. Knoppert 1970, Fast 1973). In freshwaters mixing is used mainly for the the control of nuisance blue-green algae, often causing a shift to greens, possibly by the reduction in pH associated with increased carbon dioxide concentrations and an associated activation of cyanophages (Shapiro 1973, Pastorok *et al* 1981).

Little information is available on the effects of artificial mixing on nuisance dinoflagellate blooms in saline waters. A 'Helixor' mixer (Polcon Environmental Systems) was used successfully in Sandon Dock to increase oxygen saturations and may also have caused a reduction in dinoflagellate blooms (Russell *et al* 1983), although its effects could not be separated from mussel filtration. No detailed studies of the effects of artificial mixing in saline waters have been carried out, however.

The permanent reversal of eutrophic conditions and hence reduction in algal blooms may only be brought about by strategic reductions in nutrient inputs to a given water body (e.g. Edmonson 1970, Laurent *et al* 1971). This is difficult to achieve in the South Docks as all drains and sewers have already been re-routed. The main source of external nutrients input is from the Mersey Estuary via water used to maintain water levels (top-up water). This is obviously an essential operation and the volumes involved cannot be reduced. Chemical nutrient strippers can also be used to reduce dissolved nutrient levels by flocculation and sedimentation (Wall 1971, Cooke & Kennedy 1980, Kennedy & Cooke 1980, Soltero *et al*

1981). Such applications are expensive, needing re-application when nutrients are continually added, as in the South Docks, and may not have the desired effect of phytoplankton reductions (e.g. Foy & Fitzsimons 1987).

The application of algicides to control phytoplankton blooms was once common practice in eutrophic lakes (see Landner 1976). Such solutions are short-lived and hence expensive due to the need for repeated applications. They may also be harmful to other organisms. Copper sulphate , the most widely used algicide (Luedritz 1989) can be toxic to fish below pH 7 at concentrations as low as 0.04 ppm (Tarwell 1963). For these reasons it was decided that the use of algicides was not a viable option in the South Docks.

Flushing out of algae using an external water supply has been used to reduce phytoplankton biomass in enclosed bays and in other docks (Nuttall *et al* 1989, Conlan 1989). This cannot be used in the South Docks as the only water available for flushing is from the Mersey Estuary. Use of this water would introduce large quantities of silt and sewage derived micro-organisms to the docks, this would increase the need for dredging and would jeopardize the health of watersports participants.

The direct control of phytoplankton populations by filter feeders in both fresh and marine waters has been described for a variety of species (e.g. clams, Cloern 1982, Officer *et al* 1982, Cohen *et al* 1984, Nichols 1985; *Mytilus edulis* , Loo &Rosenberg 1989, Smaal *et al* 1986; fresh water mussels, Reeders *et al* 1989; serpulid polychaetes, Davies *et al* 1989; zooplankton, Dorazio 1987 and natural assemblages of benthic species, Hily 1991). However, it was the reports of improved water clarity after increases in populations of *Mytilus edulis* in Sandon Dock, Liverpool (Russell *et al* 1983, Hawkins *et al* in press a), that raised the possibility that such a biological filter could be used to improve water quality in other Merseyside docks. Several workers have suggested that the manipulation of populations of filter feeders can be used to improve water quality (e.g. Reeders *et al* 1989, Shapiro 1990). Increased numbers of filter feeders can be brought about by direct additions of animals or encouragement of natural

settlement in the case of benthic filter feeders(Reeders *et al* 1989), or a reduction in predation in the case of zooplankton (Bendorff 1990, Langeland 1990, Moss 1990, Riemann *et al* 1990 Sanni & Waervågen 1990, Shapiro 1990). Large-scale manipulations of large bodied zooplankton filter feeders in an attempt to improve water quality has been carried out recently in fresh waters (e.g. Lynch & Shapiro 1981, Dorazio 1987, Bendorff 1990, Moss 1990). Such increases in zooplankton populations may be brought about by the removal of fish predators or provision of refuges from predation (see chapter 1). The possible use of the fresh water mussel *Dreissena polymorpha* continues to be investigated (Reeders & Vaate 1990). The effects of introductions of marine filter feeders has been studied only in small enclosures (e.g. Doering & Oviatt 1986, Riemann *et al* 1988).

The effectiveness and practicality of three approaches to control of water quality were studied, building on experiences in freshwaters and at Sandon Dock:

1) The use of artificial mixing as a method of increasing oxygen concentrations and reducing problem algal blooms.

2) The use of filter feeders (*Mytilus edulis*) to control algal blooms. Initially the effects of mussel introductions and later of natural populations were studied. Various methods of cultivation were also evaluated.

3) The evaluation and modification of water impoundment practices in an attempt to reduce the impact of ingress of water from the Mersey.

This involved the installation of a helical-type water mixer and a large population of mussels to the Graving Dock, which was used as an experimental dock due to its semi-isolated nature. Oxygen concentrations and phytoplankton populations were studied before and after mixer installation and during short term use / disuse mixing cycles. The practicality of increasing mussel filtration by encouraging natural settlement onto introduced materials was investigated. Additionally patterns of water replenishment and the direct effects of topping up were examined.

All options available for water quality management in the South Docks were then considered and evaluated and management recommendations were made to the Merseyside Development Corporation. Hence the research in this chapter is of an applied nature, with the aim of improving water quality cheaply and without interference with the day to day operations of the docks.

It was originally intended to use the Albert Dock as a control dock with a low mussel population, but this was prevented by a heavy natural settlement of mussels which occurred in autumn 1988 (see chapter 5). This meant that only changes with time could be studied. Hence the mussel populations and filtration ability of each dock were assessed at various intervals and compared to the relevant water quality data.

6.2 METHODS

6.2.1 Artificial mixing

A helical-type air lift water mixer (Martec systems 'Rotamixer') was installed in the Graving Dock in spring 1988. This was initially operated at a speed of 5000 l air hr^{-1}, but was increased to 8000 l air hr^{-1} in August 1989. Apart from short periods of breakdown and experimental shut down, the mixer was operated continuously in spring, summer and early autumn. No mixing was carried out in winter months. Actual dates of operation are given in table 6.1. Oxygen and temperature measurements at 1m depth intervals were taken as part of the hydrographical monitoring programme (see chapter 3). In July 1989 a detailed study of phytoplankton populations, with and without mixing, was carried out. Three replicate phytoplankton samples were taken every few days at 0.25, 5 and 9 m depth intervals, both with mixing and during a two week shut down of the mixer. Cell numbers of the major groups of phytoplankton were counted in each sample using the inverted microscope method (as described in chapter 3).

Table 6.1 Periods of operation / shutdown of the water mixer in the Graving Dock.

Mixer On	Mixer Off	Reason for shutdown
1-7-88	27-7-88	malfunction
5-8-88	9-9-88	malfunction
12-9-88	6-12-88	winter shutdown
20-3-89	2-8-89	experimental shutdown
9-8-89	14-11-89	winter shutdown
12-2-90	18-7-90	experimental shutdown
1-8-90	22-8-90	experimental shutdown
9-9-90	8-11-90	winter shutdown

6.2.2 Biological filtration

6.2.2.1 Introduction of mussels

A pilot stock of 600 kg of mussels, transported from the beds in the Menai Straits, was introduced to the Graving Dock in July 1988. These mussels were packed into 5 m lengths of mesh tubing (also known as Pergolari netting, supplier Kerrypack Ltd, Bristol) and suspended from a buoyed longline in the centre of the dock. This initial stock survived well and in February 1989 the introduced population was increased to 1450 kg using mussels removed from a Mersey Estuary navigation buoy during cleaning operations. In July 1989 approximately 100 mussel collector strips made of mesh tubing were suspended from the remaining space on the longline; these collected spat during the autumn settlement. Growth and survival of all mussel stocks was checked regularly throughout the period of study. In September 1990 the final total mussel population of each initial stock rope type was estimated from counts of ropes and the weight of mussels stripped from three sample ropes of each type were determined.

6.2.2.2 Estimation of filtration rates

Estimations of the size and biometry of natural mussel populations were obtained from wall samples (see section 5.3.5). Filtration rates of populations were estimated using the laboratory derived dry weight conversion of Vahl (1973) :

$$(1) \qquad P = 3.9 \cdot W^{0.60}$$

Where P is the pumping rate in lhr^{-1} $individual^{-1}$ and W is the mean dry weight of soft parts. This conversion was chosen as the experimental phytoplankton cell concentrations and temperatures used were typical of average conditions in the South Docks. For comparison, filtration rates were also calculated using the dry weight conversion of Smaal *et al* (1986), derived from measurements taken *in situ* on benthic mussels, that is

$$(2) \qquad P = 1.65 \cdot W^{0.61}$$

Additionally *in situ* filtration rates were measured for mussels from the Albert (on 8 occasions) and Queens (2 occasions). Mussels of the modal length (July 1990) for each dock (30 - 35mm for Albert Dock, 50- 55mm for Queens Dock) were placed in cages, ten mussels per cage of Albert Dock mussels and four per cage of the larger Queens Dock mussels (see Fig. 6.9). Cages were returned to the dock for two weeks before initial measurements were made. Care was taken throughout the experiments not to disturb the positions of the mussels or to break byssus threads which can effect filtration rates (Vismann 1990). Some measurements of filtration on Albert Dock mussels were carried out in the higher phytoplankton concentrations of the Queens Dock after transfer and a two week acclimatisation period. Filtration rates were measured indirectly using the particle depletion method as described by Coughlan (1969). Experiments were carried out using a series of 6, or later 8, plastic 10 l tanks, supported in a metal frame, attached to a floating pontoon (see Fig. 6.8 for experimental set-up). Five litres of dock water was added to each tank at the start of each experiment. Caged mussels were then taken from the dock, gently cleaned of debris or epiphytes and one cage placed in each of 4 or 6 replicate tanks, with two left empty as controls (Fig. 6.8). Water samples were taken after a 10 minute stabilization period and again after 40 minutes, these were preserved immediately in acidified lugols iodine and particles counted on returning to the laboratory using a ZB Coulter Counter (Coulter Counter Electronics Ltd). Mixing of water in the experimental tanks was carried out to a large extent by surface turbulence of the surrounding dock water, but to ensure complete mixing, water in the tanks were stirred manually every 10 minutes. This did not appear to disturb mussels unduly as most remained open and filtering.

The water temperature in the experimental tanks was taken on each occasion as this will effect the filtration rates (see discussion). The length of the mussels in the cages was measured regularly throughout the period of study so that any changes in filtration rate could be examined in the light of the growth of the mussels.

6.2.2.3 Experimental cultivation techniques

Several methods of collecting and on-growing naturally settling mussels within the South Docks were tried, both in suspension and over the sediment. Methods were sought which allowed survival and growth of mussels, using low cost, low toxicity materials with good spat collection properties. Primary settlement of mussels is enhanced on filamentous structures such as algae (Bayne 1964) and tufts of frayed rope are often used for collecting mussels for cultivation (Dare *et al* 1983, Hurlburt & Hurlburt 1975), but these are not suitable for on-growing and transfer to longline is required. The weave of the polyethylene mesh tubing used for on-growing of mussels in the Graving Dock (Kerrypack TM2 netting) provides a filamentous structure and it was decided to use this for both collection and direct on-growing of spat. In June 1989 approximately 100 ropes of 5m lengths of mesh were suspended under pontoons in the Albert Dock (Fig. 6.9) and from long-lines in the Graving Dock, leaving space above the sediment to prevent predation by crabs. These ropes were checked at regular intervals and the growth and survival of settling mussels was monitored.

Several methods of growing mussels on the dock bottom were investigated. In spring 1990, 60 kg of mussels from the Graving Dock were transferred to an enclosure in the Queens Dock and distributed over the sediment. These were observed by diving, after 1 week to check that they had not sunk into the sediment and again at the end of the summer to assess survival. In July 1990, 500 kg of waste scallop shells were distributed in small heaps over the sediment in parts of Queens and Salthouse Docks to provide a hard surface for settlement of filter feeders. In August 1990, experimental settlement units of rosettes of six used tyres bound by polypropylene rope were introduced to the Queens and Salthouse Docks, 5 units in each Dock. Both scallop shells and tyre units were examined by SCUBA diving in October 1990 (4 to 4.5 months after deployment) and the macroflora and macrofauna settled in thirty 10 x 10cm quadrats determined on tyre, scallop and mud surfaces. In July 1991 a further 20 tyre units were added to the Queens Dock, however these units also had a sheet of polypropylene netting (Kerrypack SD 25% density) attached to the upper surface.

6.2.3 Studies of current water management practices

These studies centred around the effects of topping-up with Mersey Estuary water and possible alternative approaches. Amounts of dissolved nutrients and sediments introduced with intake water were assessed using National Rivers Authority (NRA) data for the appropriate section of river at high tide.

The penetration and path of intake water was monitored by measuring water clarity before and after a topping-up event at various distances from the sluice gates. The use of tracer dyes would have yielded more accurate results, but was obviously not acceptable due to the high public profile and low flushing rate of the South Docks.

The amount of water entering the docks each year was assessed using electronically measured water depths taken before and after a topping-up event at the Brunswick Dock gate. Topping up procedures are outlined in Chapter 2.

One possible alternative source of replenishment water was that continually pumped out of the Mersey rail tunnel. The salinity and dissolved nutrient levels of this water were measured and the potential of this alternative was assessed.

6.3 RESULTS

6.3.1 Effects of artificial mixing

In the Graving Dock, before installation of the mixer, oxygen concentrations were typically supersaturated in the top 2m of water and decreased rapidly below this to almost zero at 4m (Fig. 6.1a). The region of maximum oxygen gradient coinciding with a temperature drop of up to 3.5 °C between 2m and 3m depth. After installation of the mixer the oxygen regime improved considerably (Fig. 6.1b), with more than 60% saturation in water above 6m

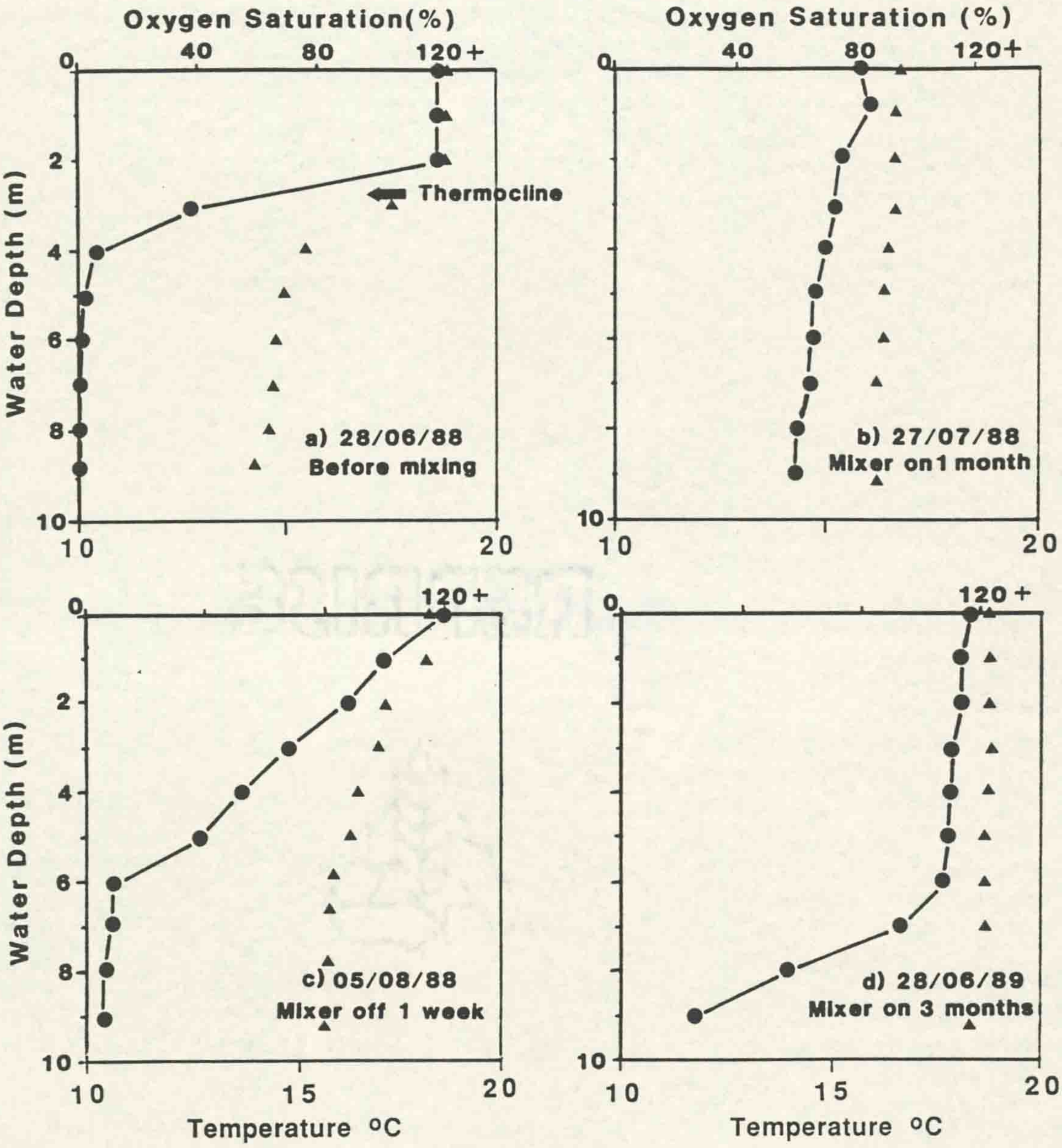


Fig. 6.1, a - d. Dissolved oxygen saturation and temperature profiles in the Graving Dock, with and without artificial mixing.

▲ Temperature °C
● Oxygen saturation (%)

occurring at all times with continual mixing. Initial oxygenation was slow, however. Thermal stratification was greatly reduced with mixing. A temperature gradient of more than 1°C per metre depth was only recorded on two occasions with mixing, and this only occurred over the surface 1m of water. When the mixer was switched off oxygen concentrations rapidly declined to reach pre-mixing levels within 1 week (Fig. 6.1c). In the hot summer of 1989 low oxygen levels were frequently seen in the deeper waters despite continuous mixing (fig. 6.1d). In summer 1990 periods of anoxic conditions were only recorded when the mixer was switched off for experimentals. In winter months oxygen concentrations remained high at all depths, without the need for artificial aeration.

Long-term variation in phytoplankton populations are described in chapter 4 and illustrated in Fig. 4.2a. The immediate effect of mixing on the phytoplankton of surface and deeper waters is illustrated in Fig. 6.2. Before the onset of mixing in the Graving Dock a dinoflagellate of the genus *Gymnodinium* was present in high densities (up to 13000 cm^{-3}) in surface waters. After mixing began these were replaced by euglenoid algae (*Eutreptiella* sp.) in densities of up to 12500 cm-3 which persisted for the rest of the summer. Euglenoid algae did not dominate at any time in following years. At 9m depth phytoplankton concentrations were much lower than in surface waters, both before and after the use of the mixer (Fig 6.2). The density of dinoflagellates (mainly *Gymnodium* sp.) showed a slight initial increase after the onset of mixing, presumably due to the transport of these phytoplankton from the surface layers. The immediate shift to euglenoid algae seen in surface waters did not occur at 9m, at this depth a mixed assemblage of low densities of dinoflagellates (*Gymnodinium* sp. and *Gyrodinium spirale*) were observed throughout the summer with diatoms becoming more important towards the autumn.

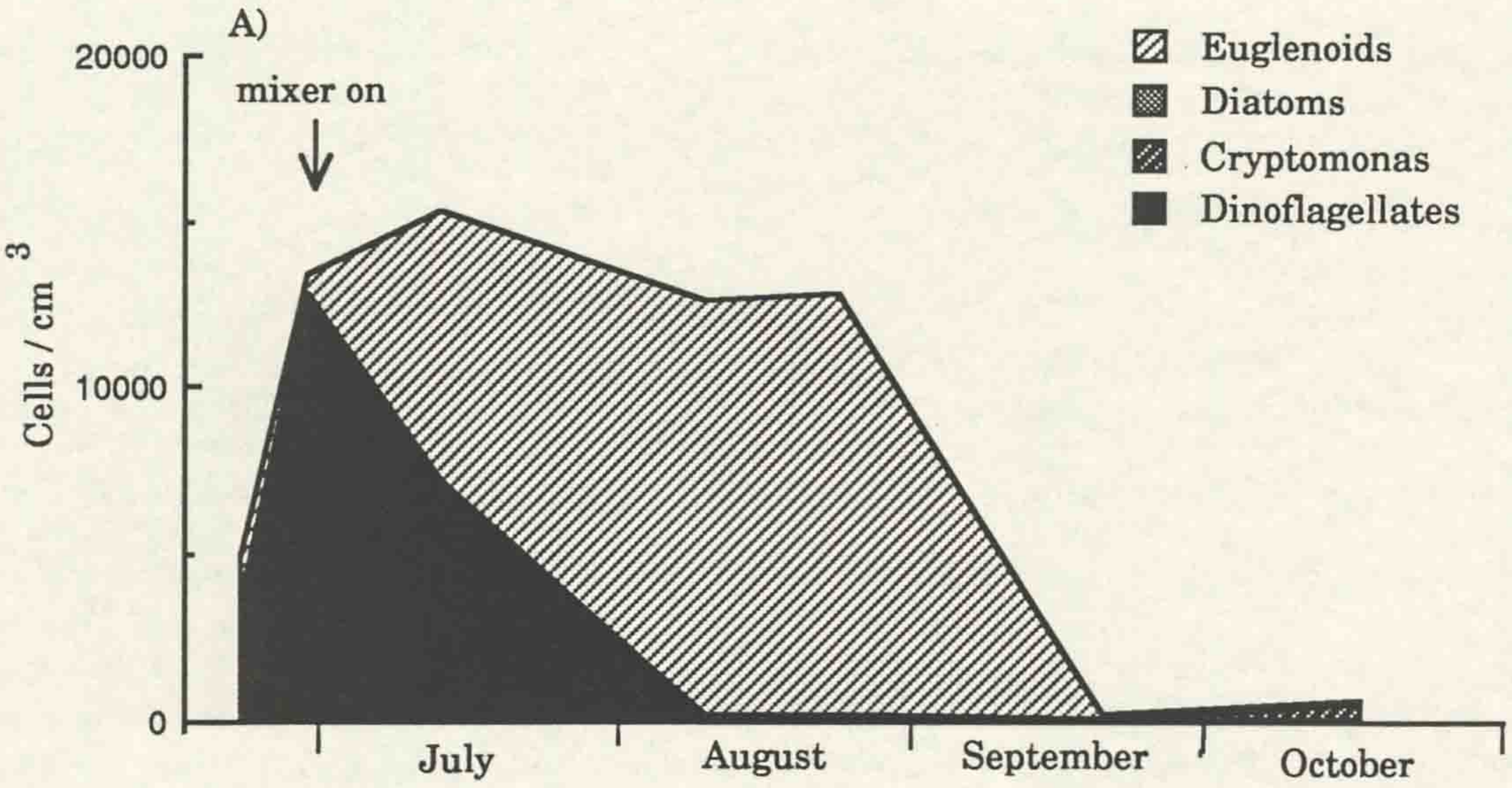


Fig 6.2 Cell concentrations of the main phytoplankton groups, before and after the start of artificial mixing in the Graving Dock, summer 1988. A) Surface, B) 9m depth

In 1989 and 1990 the annual cycle of phytoplankton followed a similar pattern to that of the Albert Dock with a spring diatom/*Phaeocystis* peak followed by mixed dinoflagellate/ *Cryptomonas*/diatom populations in summer. The very dense summer dinoflagellate concentrations seen in Queens Dock in 1988 and 1989 did not occur in either the Graving or the Albert Docks (see Fig. 4.2).

When the mixer was turned off in July 1990 in order to study short term changes in phytoplankton, slight thermal stratification developed which gradually deepened as time progressed (Fig. 6.3a). However, after two weeks without mixing the maximum temperature gradient was only 0.7°c, between 3 and 4m depth, despite the hot, sunny and still weather (see chapter 2) Oxygen concentrations showed much greater change with less than $1mgl^{-1}$ occurring below 6m depth after two weeks without mixing (Fig. 6.3b). After resuming mixing for one week the water was restored to a fully mixed condition with very similar oxygen and temperature levels at all depths. The total numbers of phytoplankton cells in surface waters increased immediately after mixing was stopped, due to a rise in numbers of micro-flagellates, and then decreased again (Fig. 6.4 & 6.5). A second increase occurred immediately after mixing was resumed. Phytoplankton densities were higher in surface than deeper waters with mixing. This situation became much more pronounced when the mixer was turned off and phytoplankton numbers in surface waters increased approximately fourfold. No sustained shift in species composition occurred with differences in mixing, although a slight rise in dinoflagellate and μ flagellate numbers was seen immediately after mixing ceased and a small increase in diatoms occurred when mixing was resumed (Fig 6.5). The variations seen in phytoplankton cell numbers and species present are very small compared to possible changes of several orders of magnitude seen over similar time scales with constant mixing or non-mixing regimes.

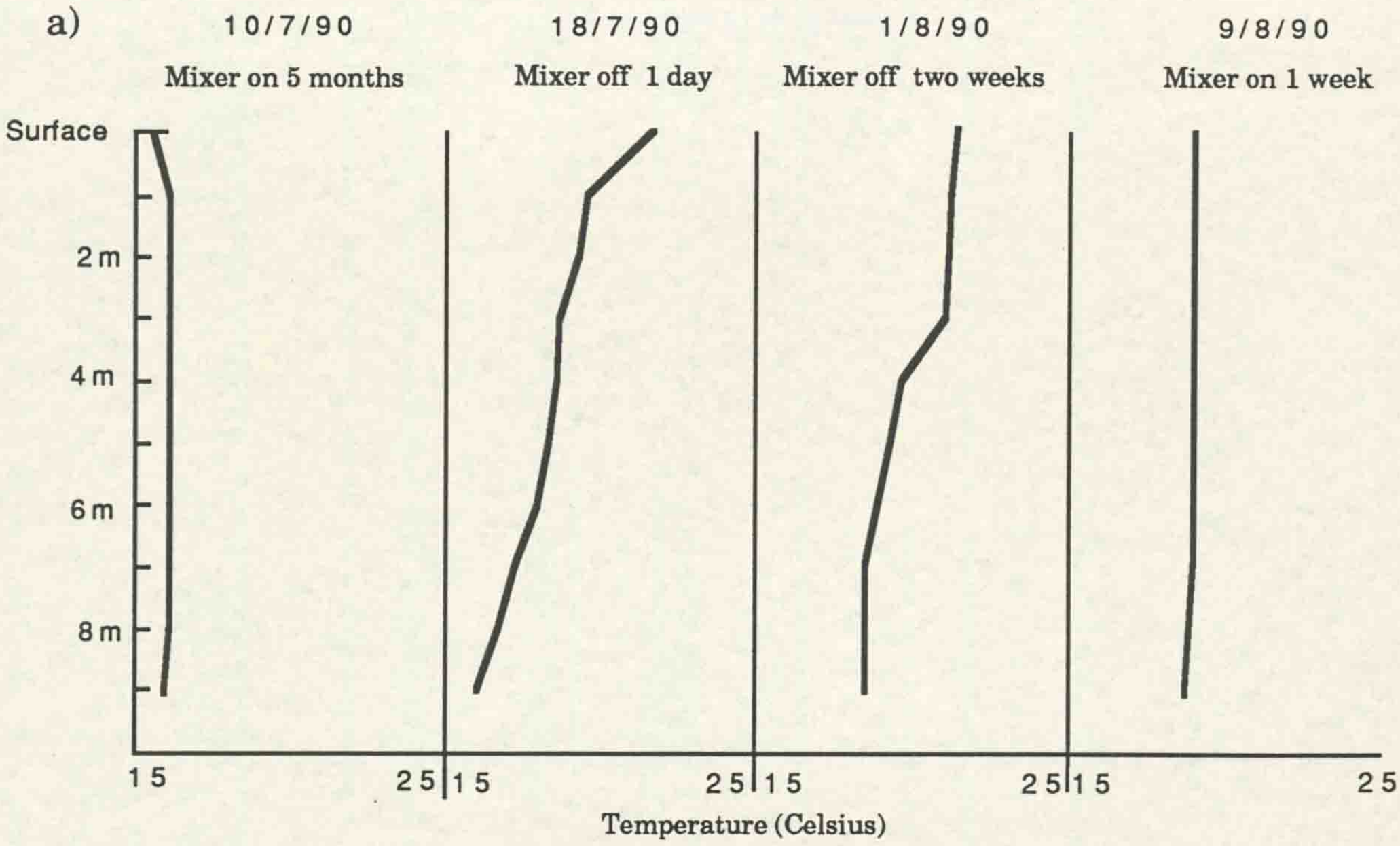


Fig 6.3 Temperature (a) and oxygen (b) profiles in the Graving Dock, with and without mixing during the period of detailed phytoplankton study, summer 1990.

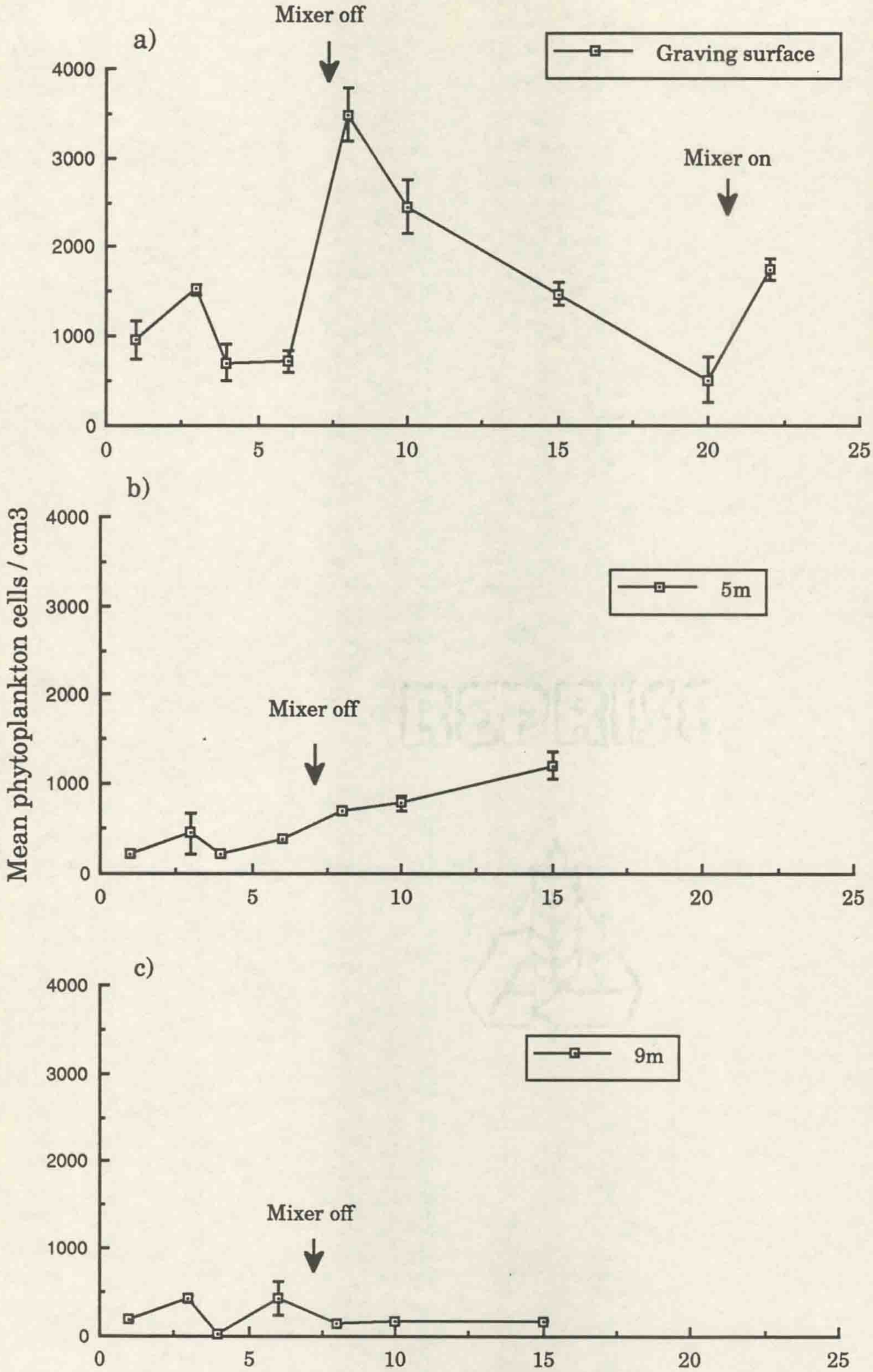
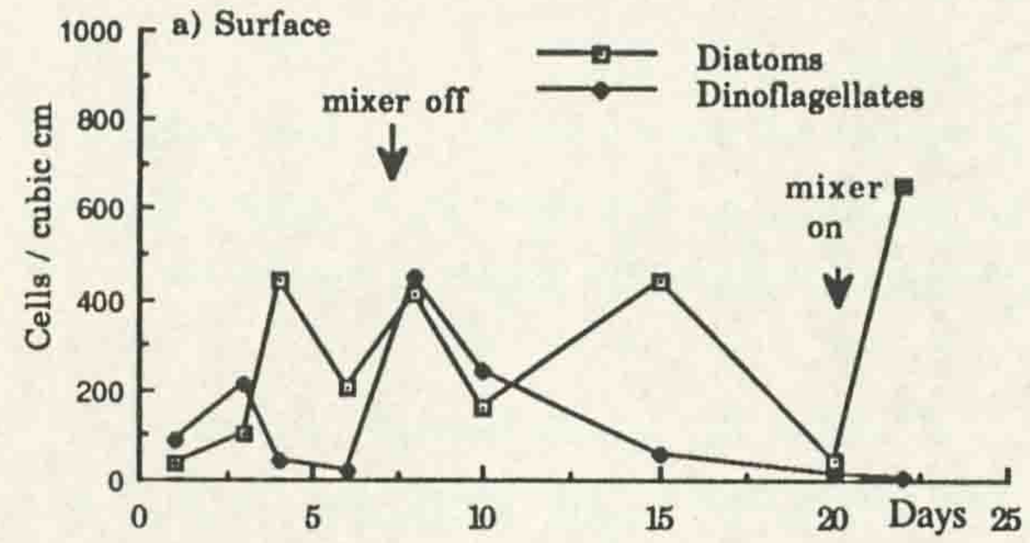
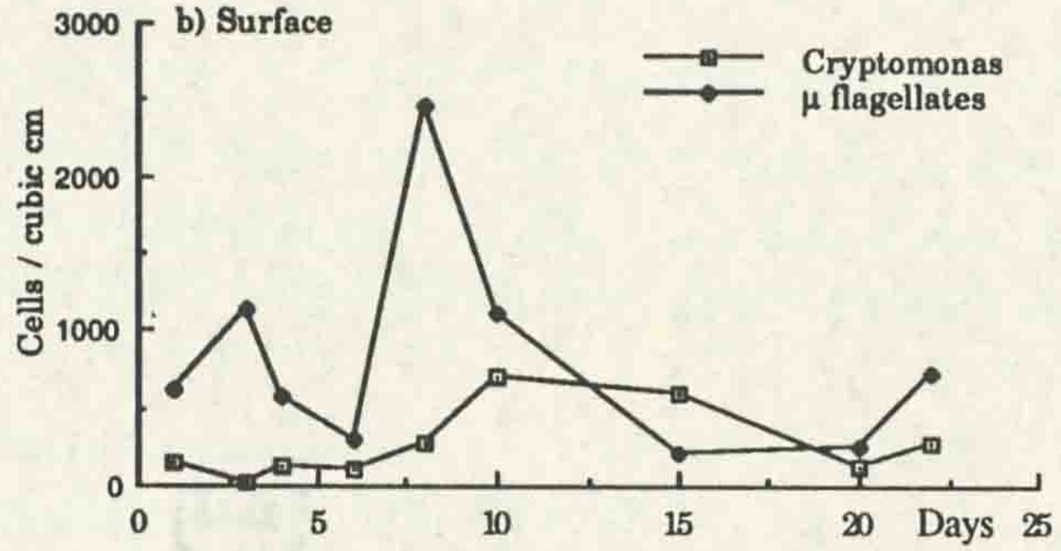
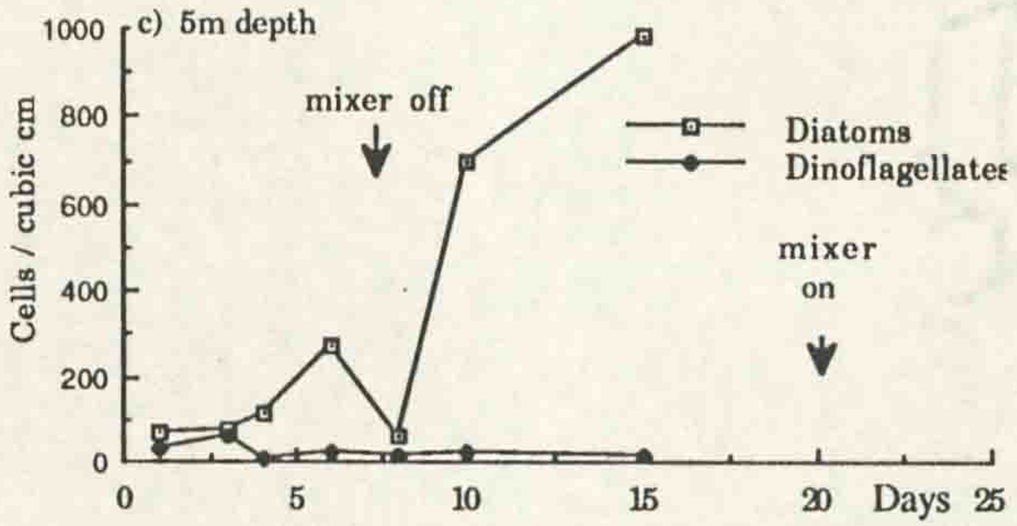
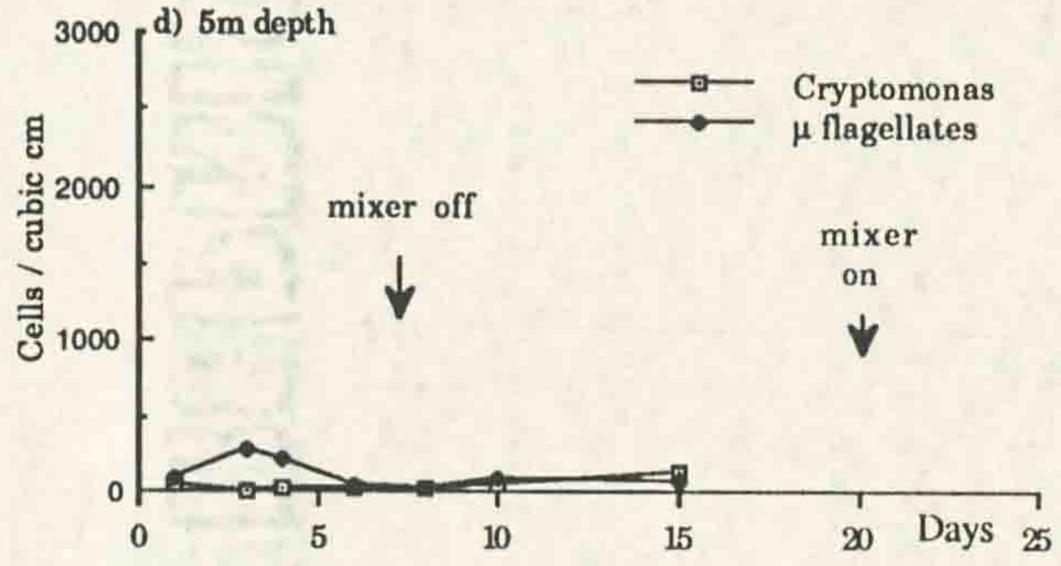


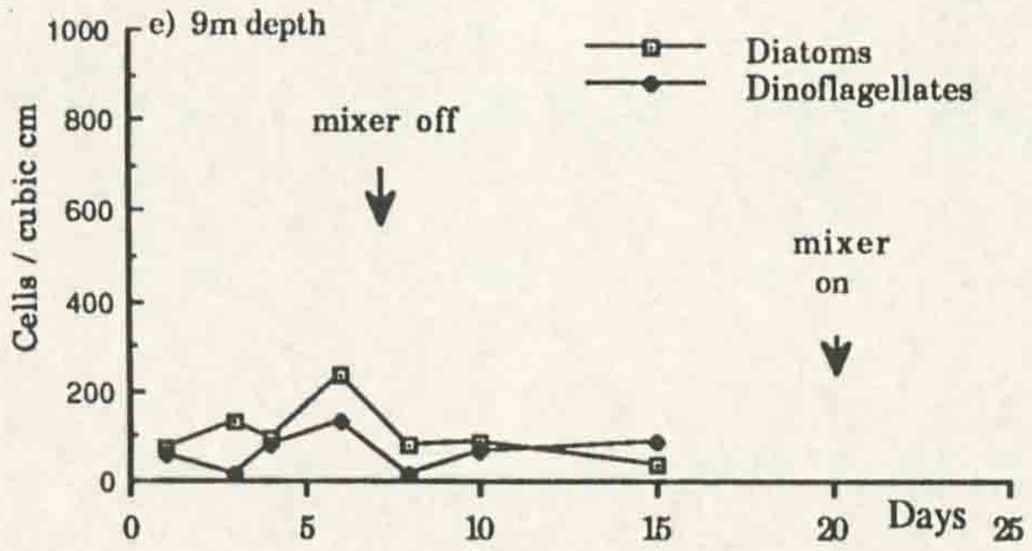
Fig. 6.4 Concentrations of phytoplankton cells with depth in the Graving Dock, with and without water mixing. Means of three replicate samples ± S.E. Missing values are due to failure of the sampling bottle.











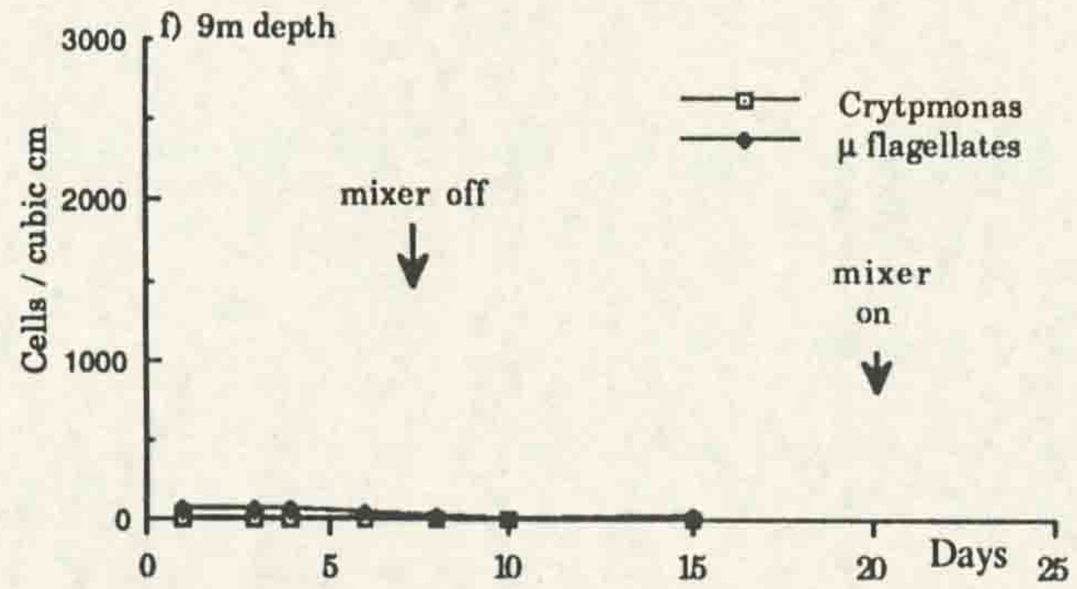


Fig. 6.5 a to f. Cell concentrations of main phytoplankton groups,
with and without mixing at surface, 5m and 9m depths.
Diatoms & Dinoflagellates (a, c, e) and Cryptomonas & µ flagellates (b, d, f).

6.3.2 Biological filtration

The estimated total filtration rates of mussel populations of each dock, calculated from the mean dry weight are shown in Table 6.2. This includes both natural and introduced populations. An indication of the relative filtration of water in each dock is given by the dock volume filtration time. This is the theoretical time taken for one dock volume of water to pass through the resident mussel population. The actual time taken for mussels to filter all the water in the dock is of course much greater due to refiltration effects. Refiltration of water will occur even in a completely mixed water body because filtered water is not partitioned away from unfiltered water. However, refiltration will be even greater if the water is poorly mixed. The figures of mussel filtration rates derived from laboratory measurements (from Vahl 1973) are not necessarily a good reflection of natural filtration rates in the field (Bayne *et al* 1989, Widdows *et al* 1979a). Estimates made using conversions derived from *in situ* measurements (Smaal *et al* 1986) which are likely to be more representative are considerably lower (Table 6.3). Both these estimates use conversions from dry weights which may introduce errors due to large seasonal variations in condition and gonad state. Values given in the table should therefore be regarded as relative values only. These figures show that mussel filtration was initially very low, but gradually increased in all three docks with relative filtration potential in 1989 and 1990 being in the order Albert > Graving > Queens Docks (Table 6.2). A more detailed comparison of mussel populations in summer 1990, their calculated filtration rates and dock size is given in Table 6.3, calculated figures for Sandon Dock are also included based on mussel populations given in Hawkins *et al* (in press a) and Naylor (1983).

Mean filtration rates as calculated from *in situ* measurements are also given in brackets in Table 6.3 for comparison and are lower than calculated figures. Filtration rates measured on each occasion are shown in Fig. 6.6.2. Variability between replicates on each occasion

Table 6.2 Estimated filtration times of docks by resident mussel populations, summers 1988, 1989 and 1990. Filtration rates calculated from Smaal *et al* (1986)

	Dock Volume Filtration Time (Days)			
	1988	1989	1990	Water mixer.
Graving Dock	19.0	6.7	3.9	Yes
Albert Dock	Negligible filtration	2.2	1.5	No
Queens Dock	Negligible filtration	8.9	4.8	No

Table 6.3. - Mussel populations and relative filtration potentials in the South Docks (August 1990) and Sandon Dock, Liverpool.

DOCK	Dock Volume $m^3 \times 10^3$	Mussel Nos m^{-3} water	Mean mussel length (mm)	Mean dry weight soft parts (g)	Weight specific filtration rate l/hr/mussel	Dock vol. $filt^n$ time (days)
GRAVING	54.0	12	40.6	0.348	**0.87** 2.07	**3.9** 1.6
ALBERT	154.7	36	41.3	0.281	(0.50) **0.76** 1.82	(2.3) **1.5** 0.6
QUEENS	276.3	5	50.1	1.127	(1.18) **1.77** 4.2	(3.2) **4.8** 2.1
SANDON *	430.0	11		1.2	**1.8** 4.25	**2.1** 0.9

*Data from Naylor (1983) & Russell *et al.* (1983).
Weight specific filtration rates and Dock volume filtration times are calculated from Vahl (1973) (plain text) and Smaal *et al* (1986) (in bold). Values in parentheses are from *in situ* measurements of filtration (Albert n = 40 determinations over 8 sampling occasions, Queens n = 7 over two occasions).

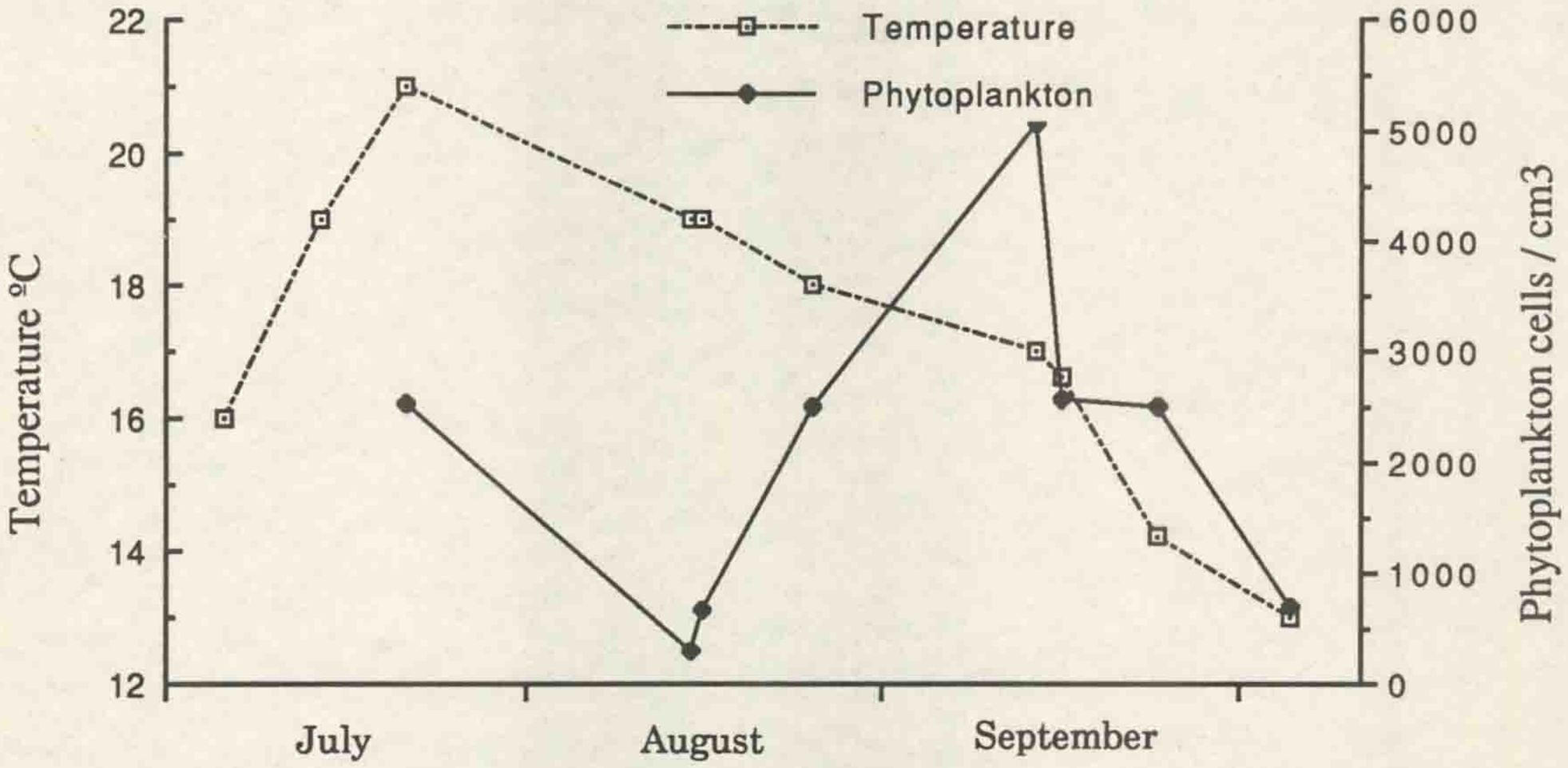


Fig. 6.6.1 Temperature and phytoplankton concentrations present during filtration rate experiments. Note missing phytoplankton values on 2 dates.

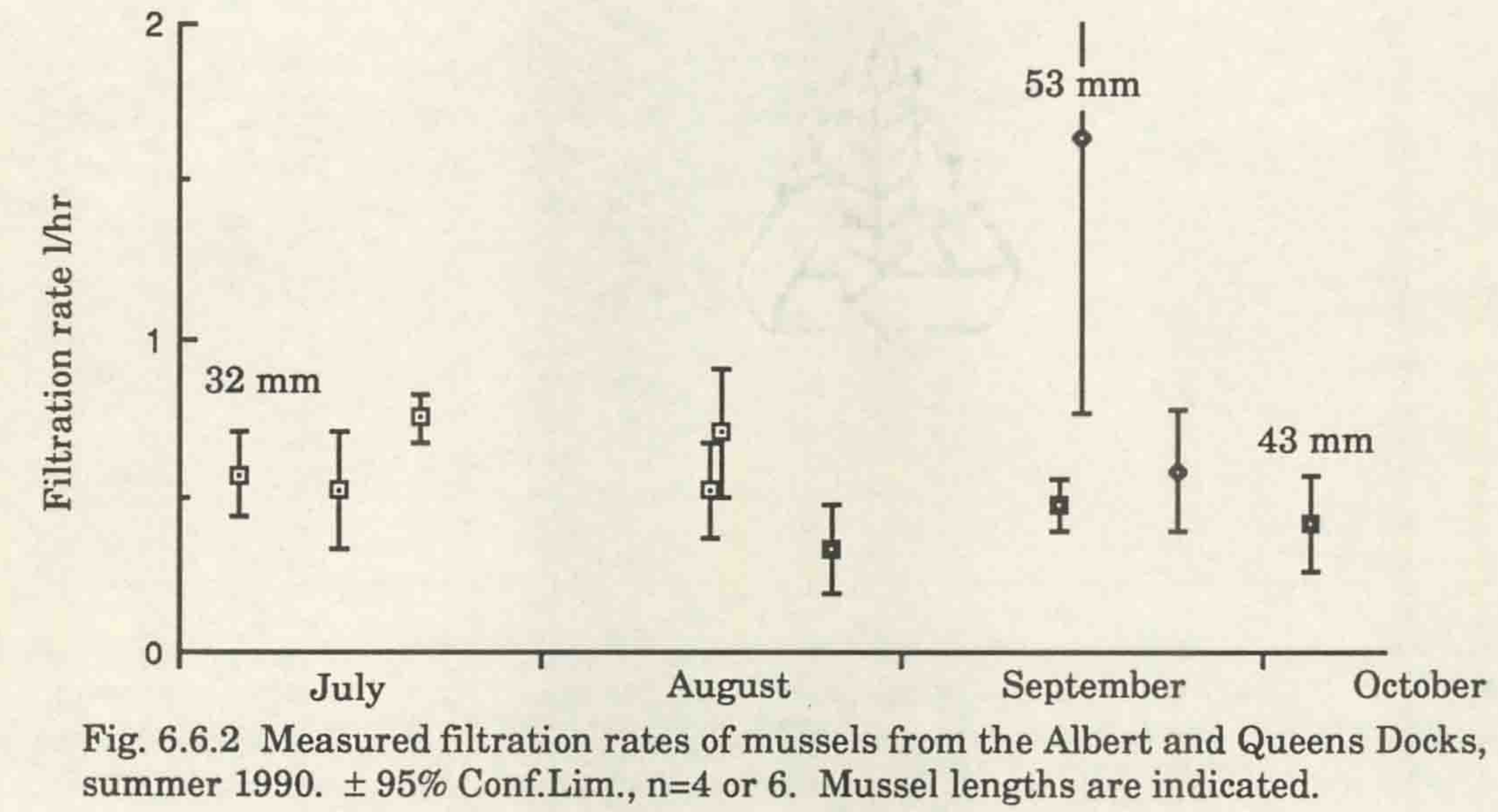


Fig. 6.6.2 Measured filtration rates of mussels from the Albert and Queens Docks, summer 1990. ± 95% Conf.Lim., n=4 or 6. Mussel lengths are indicated.

▫ Albert Dock mussels in Albert Dock, ▪ Albert Dock mussels in Queens Dock,
◆ Queens Dock Mussels in Queens Dock.

Table 6.4 Secchi disc extinction depths in Graving, Albert and Queens Docks in summers (June - August) of 1988, 1989 and 1990.
Kruskal-Wallis statistics for comparisons for each dock between years (across) and for each year between docks (down) are given. *** = significant (P<0.001). N.S. = not significant (P>0.05).

		1988	1989	1990	Kruskal-Wallis statistic (H)
Graving Dock	Median	1	2.5	4.5	25.65
	Range	0.6 - 1.5	1.2 - 4.0	4.0 - 7.0	***
	n	12	11	8	
Albert Dock	Median	1.1	3	4	15.86
	Range	0.8 - 1.5	2.2 - 4.5	2.5 - 4.5	***
	n	7	11	8	
Queens Dock	Median	1.2	1	1.5	3.59
	Range	0.9 - 1.4	0.5 - 2.0	0.9 - 2.5	N.S
	n	6	11	8	
Kruskal-Wallis statistic (H)		3.39 N.S	22.13 ***	16.74 ***	

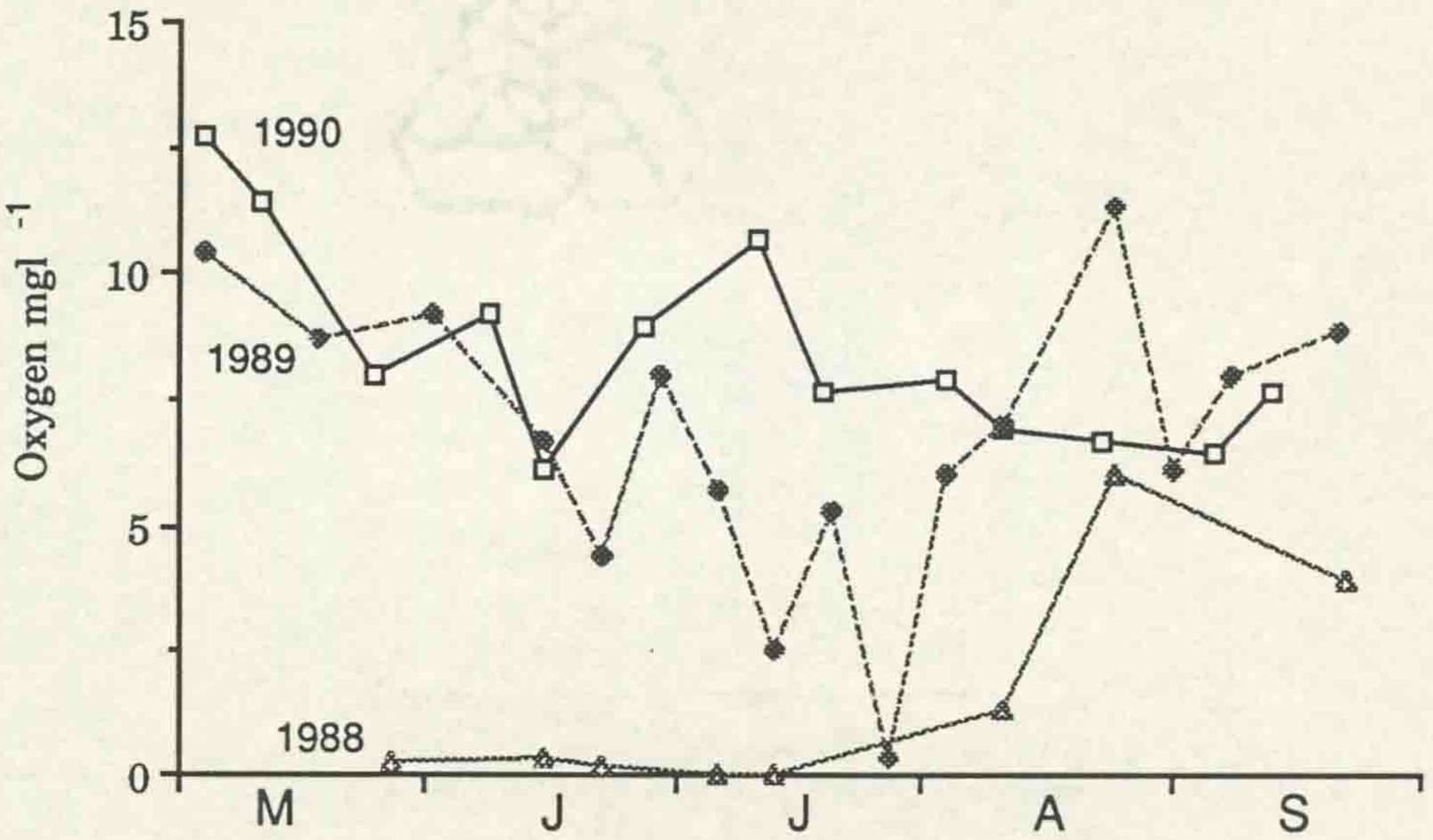


Fig 6.7 Oxygen concentrations at 5m depth in the Albert Dock, summers 1988, 1989 and 1990.

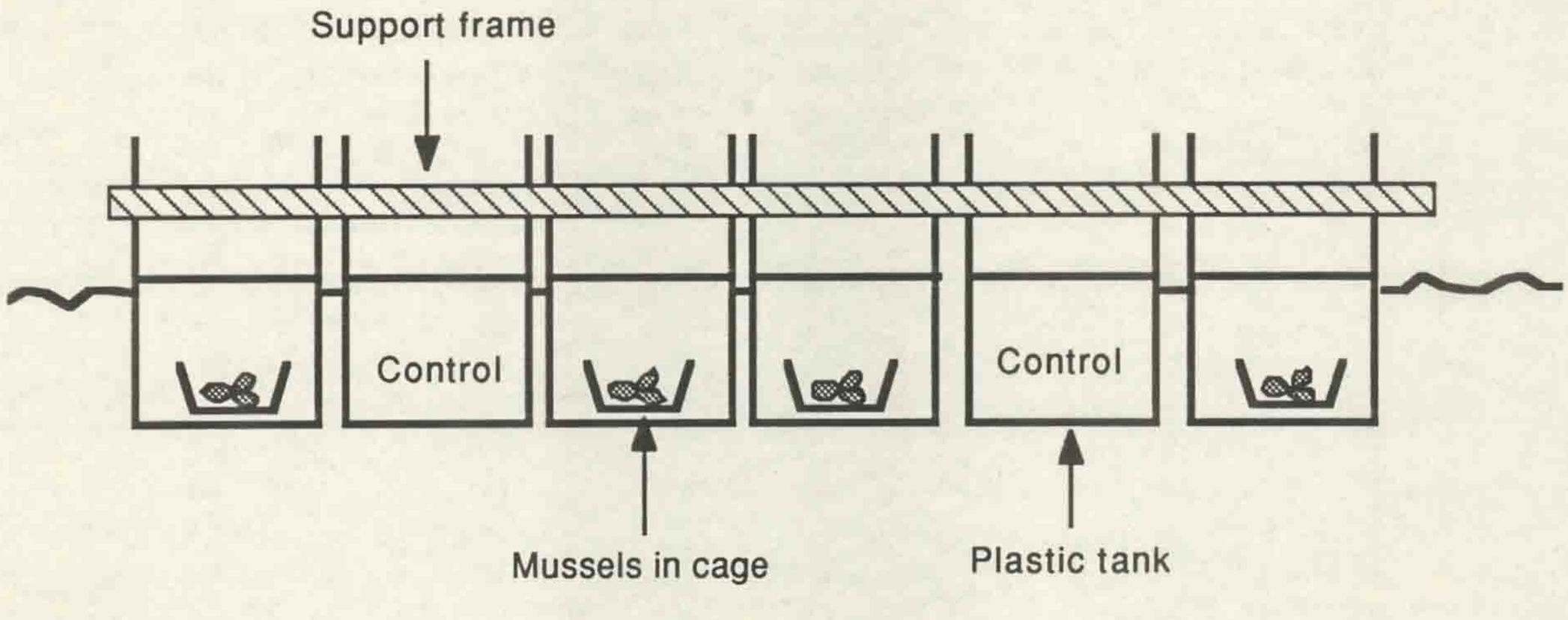


Fig 6.8 Experimental set-up for measurement of in situ filtration rates.

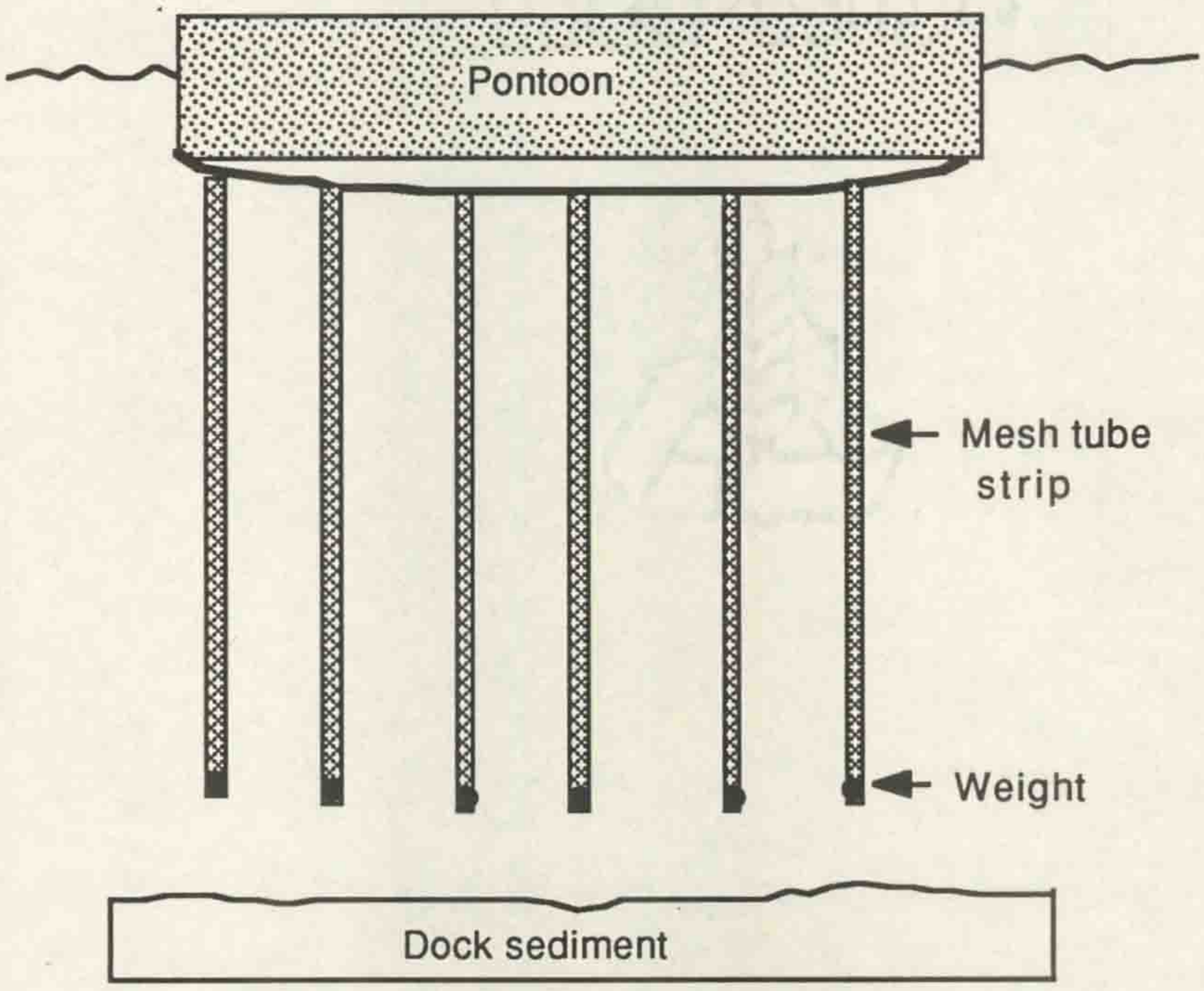


Fig 6.9 Arrangement of mesh collection ropes.

generally low, with the notable exception of the first measurement made on mussels from Queens Dock. Phytoplankton populations on this day consisted mainly of *Skeletonema* chains which may have increased errors in counting as the Coulter Counter is designed for counting spheres rather than elongated shapes. Variation between sampling dates was generally low for Albert Dock mussels with little obvious change with temperature, phytoplankton or growth of the mussels (Fig 6.6.1 & 6.6.2).

Median summer water clarity and the significance of differences between years / between docks are given in table 6.4. The Kruskal-Wallis test was used as a conservative non-parametric test. The variances were unlikely to be homogenous so ANOVAR was not applicable. The repeated use of one way comparisons in this way increases the chance of type 1 errors, therefore a stringent probability level of $P<0.01$ was applied for rejection of the null hypothesis of no difference. In practice all significant differences were at the $P<0.001$ level. In 1988 all docks had similar water clarities (Table 6.4), but in 1989 and 1990, after mussel settlement or introduction, marked differences were seen with Albert and Graving Docks having greatly improved water clarity. In Queens Dock no significant difference between years was seen. As described in Chapter 5 chlorophyll *a* measurements indicate that increases in water clarity were due to decreases in phytoplankton biomass. Maximum summer (June - Aug incl.) chlorophyll a concentrations were 34, 17, & 10 μgl^{-1} in the Albert Dock in 1988, 1989 and 1990 respectively while in Queens Dock the corresponding values were 30, 37 and 33 μgl^{-1}. Secchi Disc extinction depths were used for comparisons as these were measured more frequently.

Decaying phytoplankton blooms are a likely major contributor to biochemical oxygen demand (BOD) (see chapter 3). Therefore another potential benefit of increased filter feeding is the decreased occurrence of low oxygen concentrations. In the Graving Dock it is impossible to relate this to mussels as a mixer was also used. In the Albert Dock the occurrence of low oxygen bottom waters in summer was greatly reduced between 1988 and

Table 6.5 Settlement of macrofauna and macroflora on introduced and natural settlement surfaces in the South Docks. Scallops introduced July 1990, tyres Aug. 1990, mesh ropes June 1988.

		Mytilus per m^2	Molgula per m^2	Botryllus % cover	Bryozoan % cover	Hydroid % cover	Vaucheria % cover
	Mud	0	80	0	0	0	0
Queens Dock Oct-1990	Shells	0	30	0	0	<5	0
	Tyres	0	80	0	<5	<5	0
	Mud	0	<10	0	0	0	40
Salthouse Dock Oct-1990	Shells	10	290	< 5	0	0	20
	Tyres	0	0	0	< 5	< 5	0

		Number of Mytilus / m of rope	Mean mussel length mm
Albert Dock Nov-1989	Mesh rope	580	25
Graving Dock Nov-1989	Mesh rope	60	19

1989 with further improvements in 1990 (Fig. 6.6). Oxygen saturations of less than 20% were recorded for a maximum duration of over 2 months in 1988, for 2 - 3 weeks in 1989, with no occurrence in 1990.

6.3.3 Experimental cultivation techniques.

Mussels transferred onto the sediment in the Queens Dock enclosure were found to be alive and lying on the surface of the sediment on first examination. By the end of the summer, however, all mussels lying on the sediment were dead. Only those individuals that had become attached to debris elevated above the sediment had survived.

The success of the various experimental settlement devices is illustrated in Table 6.5. Mussels settlement on mesh ropes was much more dense in the Albert than the Graving Dock, and settled mussels were of a greater mean length by the end of November. Differences in density were caused by variations in settlement rather than interspecific competition as the mesh surface between mussels in the Graving Dock ropes was not colonised by other species. In the Albert Dock mussels settled on the ropes in September after previous colonisation by firstly hydroids and later *Molgula manhattensis*, with almost complete cover of each species at each stage.

Mussel settlement on tyres and scallop shells on the Dock bottom was very poor (Table 6.5). No mussels settled on either substrate in Salthouse Dock and only a couple of specimens were found on scallop shells in Queens Dock, although mussels had settled on the walls of the Docks during the same period (see chapter 5). The settlement of *Molgula manhattensis* on scallop shells in the Salthouse Dock was better than on sediment alone. This was not the case in the Queens Dock where *Molgula manhattensis* was seen to attach directly to the sediment, especially on more compacted raised areas. No detailed study of the mesh and tyre units was carried out as these were installed at the end of the project, however, a brief inspection by sports divers showed some filter feeder cover on these structures within a few months.

6.3.4 Water Management

A gradual increase in the total volume of water needed for topping up was seen from 1988 to 1990 (Table 6.6). However, changes in management practices, with more frequent but smaller intakes of water, reduced the mean depth of water addition from 0.18 m in 1988 to 0.15 m in 1990. The amount of suspended solids and nutrients brought in to the Docks will have increased with increased water intake (Table 6.6). Figures given for intakes of nutrients and solids are based on NRA data from the Mersey for 1989 as these were the only figures available. Concentrations of these substances in the Mersey are unlikely to vary a great deal from year to year.

Measurements of water clarity just after water intake (Fig 6.10) show that penetration and immediate effects of such water is limited to the first two docks of the chain (Brunswick and Coburg).

Analysis of water pumped from the Merseyrail tunnel, carried out both by myself and an outside consultancy (data obtained from MDC), showed that while suspended solids were lower in tunnel water than in Mersey water (mean 47 mgl^{-1} compared to 67 mgl^{-1}) all measured dissolved nutrients were higher (total inorganic nitrogen > 5 mgl^{-1} in tunnel, 1.0 mgl^{-1} in Mersey, orthophosphate 0.38 mgl^{-1} in tunnel, 0.18 mgl^{-1} in Mersey), although particulates are not considered. Salinity of the tunnel water was lower than that of the dock with a mean of 18 °/oo compared to 26 °/oo in the Docks.

Table 6.6 Water intake to the South Docks during Jan. 1988 to Dec. 1990. Estimates of nutrient and sediment load of this water are made using NRA data for high tide at Tranmere sampling station on the Mersey in 1989.

	1988	1989	1990
Mean depth of water intakes (m).	0.18	0.173	0.15
Number of intakes.	28	38	49
Total volume of water intake over year (m^3).	1.33×10^6	1.79×10^6	2.01×10^6
Annual Tot. inorg. nitrogen intake (kg)	1350	1825	2045
Annual orthophosphate intake (kg)	239	322	361
Annual susp. solids intake (kg)	8.9×10^6	12.0×10^6	13.4×10^6

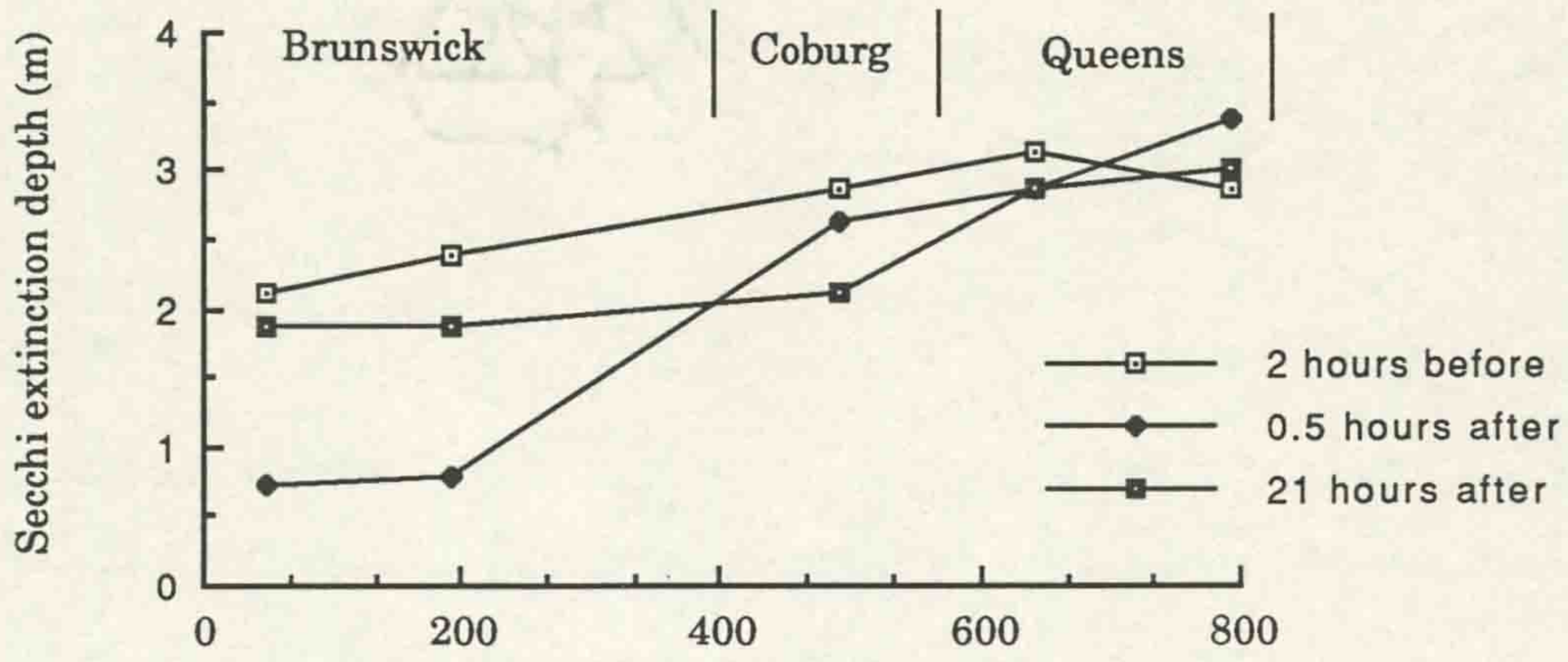


Fig 6.10 Effect of intake of Mersey water on the water clarity of adjacent docks, before and after time of intake

6.4 DISCUSSION

6.4.1 The effects of mixing in the Graving Dock

The continued occurrence of low oxygen levels in deeper waters of the Graving Dock indicates that the mixer was under-powered for the size of the dock and that mixing was incomplete. However, the virtual elimination of thermal stratification suggests that the deficiency was marginal. After early August 1989, with the increase in mixer speed, mixing appears to be more complete and the occurrence of low oxygen conditions only occurred when the mixer was turned off (Fig 3.1). The development of anoxic conditions without associated thermal stratification indicates a high oxygen demand from the sediments requiring a fast turn over of water to prevent oxygen depletion. As the sediment continued to experience prolonged periods of anoxia no colonisation by benthic fauna was seen. The increased depth of well oxygenated water did, however, allow penetration of fauna on the walls into deeper areas (see chapter 5).

Several studies in freshwater lakes reported an immediate increase in clarity due to distribution of surface concentrations of phytoplankton throughout a greater water volume (e.g Haynes 1973). This did not occur in the Graving Dock due to incomplete mixing and the continued concentration of phytoplankton in surface waters (Fig 4.1) and possibly the suspension of sediments from around the mixer.

The long term direct effect of mixing on total phytoplankton biomass is difficult to assess due to the effects of mussel filtration, but the failure of mixing to reduce phytoplankton biomass in summer 1988, when mussel filtration was low, suggests destratification alone played a small part in improvements seen in later years. In studies of 40 artificially mixed freshwater lakes a decrease in phytoplankton biomass was seen in only 57% of lakes in which complete mixing occurred. In those lakes where mixing was incomplete, however, phytoplankton

generally remained at similar levels or increased (Pastorok *et al* 1980). As mixing in the Graving Dock was incomplete (in terms of oxygen concentrations) it is perhaps not surprising, therefore, that no direct reduction of phytoplankton biomass by mixing was observed.

The persistence of much higher algal concentrations in surface than deeper layers also demonstrates that mixing was incomplete. If mixing were effective an homogenous distribution of phytoplankton throughout the water column would be expected. If the mixer is not powerful enough to transfer phytoplankton cells to depths where light intensities are lower, little effect on phytoplankton biomass can be expected.

High dinoflagellate biomass (>5 mg wet weight l^{-1}) was associated with strongly stratified conditions and high water temperatures in the Albert and Graving Docks. These blooms did not occur when the water column was mixed as a result of either natural conditions or artificial mixing (see figs 3.1a, 3.2a & 4.2a,b). Consequently no such dense populations were seen in the Graving Dock after the onset of mixing, or in the Albert Dock in 1989 or 1990 when pronounced thermal stratification did not develop in months with high water temperatures (July / August). In the Queens Dock thermal stratification was not necessary for the development of dinoflagellate blooms which developed in July / August 1989 and 1990 as water temperatures rose above 15 ^{o}C (fig 3.3a & 4.2c). In the Graving Dock the switch from dinoflagellates to euglenoid algae occurred with mixing at a time when mussel populations were low. These results indicate that artificial destratification alone is of use for controlling dinoflagellate blooms in deeper docks, thus confirming observations made by Russell *et al* (1983), but not in shallow docks where such growths may develop in mixed conditions.

The switch from *Gymnodinium* to *Eutreptiella,* seen in the surface waters of the Graving Dock after mixing began may have been brought about by the supply of organic material from the dock sediments that favoured the growth of these heterotrophs. Alternatively *Eutreptiella*

may have been better able to resist being transported out of the photic zone. Certainly, the dense *Eutreptiella* bloom whichdeveloped at the surface was not reflected in phytoplankton populations at 9m depth (Fig. 6.2).

It is unlikely that even higher powered mixing would be able to reduce the *Phaeocystis* / diatom blooms which occur in spring as these are present under natural conditions of strong mixing both in the South Docks and the Mersey Estuary (A. Jemmett unpub.). Such blooms are a lesser problem than dinoflagellates because, although they discolor the water, they do not contain potentially toxic species.

The small magnitude of changes in structure of phytoplankton populations seen during the detailed short term study with and without mixing was not surprising, given that the thermal stratification which developed was slight. In order to properly determine the effects of mixing on marine / coastal phytoplankton further studies with repeated experiments producing complete mixing, with adequate controls and without other complicating factors such as dense mussel populations is required. Natural differences between adjacent docks illustrate the need for well matched controls in such studies. The large variations in phytoplankton abundance and dominant species from year to year also illustrate the problems involved with the common practice of using 'control' years in mixing studies carried out on one waterbody alone (e.g.Fast 1973, Ellis & Tait 1981, Bailey-Watts et al 1987).

One potential benefit of mixing is the lowering of dissolved nutrient concentrations. Release of inorganic phosphorous and nitrogen from the sediments is greater under anaerobic conditions (Mortimer 1971, Hallberg *et al* 1976). Also in anoxic waters reduced forms of nitrogen predominate and the supply of nitrate for denitrification, and hence loss of N to the atmosphere, is diminished (discussed in chapter 3). The possible effects of mixing on phosphorus release are reviewed in Pastorok *et al* (1980. Opinion is often conflicting,because, although sediment release is decreased under aerobic conditions the increased flow over sediments, raised hypolimnetic water temperatures and increased biological perturbation

associated with mixing may actually result in an increase in phosphate release. Again it is likely that incomplete mixing, as occurred in the Graving Dock is the worst possible situation with high nutrient release in anoxic but warmer bottom waters and increased supply of released nutrients to upper layers. Slight differences in dissolved nutrient concentrations were seen from year to year, but these cannot be related to oxygen concentrations or mixing due to a multiplicity of other possible influences.

Although a direct reduction of total phytoplankton by mixing cannot be relied on, mixing may help to reduce phytoplankton blooms when combined with biological filtration. Mixing will increase water flow over the beds of filter feeders and help to reduce refiltration of water.

The environmental problems associated with mixing are few, so high installation and running costs are normally the main deterrent to installation of mixing systems in redeveloped docks. Helixor type mixers, which may protrude several metres above the sediment surface cannot be used where this would provide a hazard to yachts and pleasure boats. Noise levels around dock-side compressors may be unacceptable in areas used for housing, business or recreation. Mixer types with visible moving parts above water may be unacceptable for aesthetic reasons. Finally, mixer breakdown is not uncommon and should this occur in a dock in which the fauna has become dependent on artificial mixing for survival, mass mortalities of benthic fauna and fish could occur, unless an appropriate strategy for use in emergencies is adopted.

6.4.2 The influence of biological filtration on water quality in the South Docks

6.4.2.1 Measured and estimated filtration rates in the South Docks

An evaluation of the possible effects of filtration by mussels in the South Docks depends to some extent on the determination of filtration rates at any one time with reasonable accuracy. Filtration rates in mussels are affected by a range of environmental factors, of

which temperature and food availability are likely to be of most importance in the South Docks. The effects of temperature and suspended particle concentrations on filtration are the subject of many experimental studies (e.g. Tenore & Dunstan 1973, Widdows 1973, Schulte 1975, Bayne *et al* 1976, Widdows *et al* 1979a, Riisgard & Randlov 1981) Reports from various laboratory studies examining these variables over a narrow experimental range may give conflicting results as to the occurrence or direction of any effects, but this is clarified in a review by Winter (1978). The rate of filtration increases with increasing temperature up to an optimum temperature (between 15 and 20 ^{o}C) where filtration rates level out. With any further increase above the optimum range (at around 25 to 30 ^{o}C) the filtration rate decreases drastically. Such high temperatures are never reached in the South Docks. From a low threshold food concentration, filtration rate increases rapidly with rising concentrations, quickly reaching a plateau, however, above a certain level filtration rates decrease steadily as food concentrations rise.

The filtration rate (F) of a mussel increases with increasing body size (as dry weight of soft tissues, W) according to the general empirical relationship $F = aW^{b}$ (Winter 1978). Most of the values obtained for b are between 0.66 and 0.82 indicating a relationship somewhere between surface area (0.67) and body weight (1). However, all such estimates are based on a mean filtration rate over a certain time period. If multiple measurements are made on animals of different sizes and the maximum filtration rates for each are used, filtration rates correlate more closely with surface area (Jones *et al* 1992). This would perhaps be expected as filtration ability depends on the gill area available. The closer relationship of filtration rates to the square of linear dimension, rather than weight gives further reason for the use of lengths rather than dry weights in the calculation of size-specific filtration rates. Furthermore, large seasonal fluctuations in weight due to changes in condition and gonad size confound the use of weights. In the absence of suitable published conversion equations relating to length, however, it was necessary to use dry weight conversions as described.

It is clear that factors such as mussel size, temperature and food concentrations must be taken into account when estimating filtration rates of mussel populations and dock volume turn over times. Regular *in situ* measurement of filtration rates of the resident mussel population, as carried out in the Albert and Queens Docks are the best estimate of the filtration ability of a mussel community at any one time. Comparison of figures obtained from literature and measured values illustrate this well. Measured filtration rates (Table 6.3) for the Albert Dock mussels were only 27% of the rates estimated from laboratory derived figures (Vahl 1973) and 65% of *in situ* rates measured in bottom cultivated mussels (Smaal *et al* 1986). The rates measured by Smaal and co-workers were carried out using a submerged flow-through tunnel system in which no movement of the mussels took place. It is possible that transferral of mussels from suspension in the water to experimental tanks may have caused some disturbance and reduced the filtration rates measured in mussels from the South Docks. The accuracy of filtration rate calculations from literature values is not an important issue when comparative rates between docks or years are required, as proportionally similar differences are indicated by each method of estimation (Table 6.3). However, filtration rates measured under natural conditions are essential for quantitative estimates of filtration and water turnover times.

Low variation in filtration rates measured in Albert Dock mussels over summer 1990 was observed despite growth of mussels and large changes in temperature, food concentration and food type (Fig 6.6.1 & 6.6.2). This may be due to factors acting in opposition over this period, with high temperatures, low phytoplankton concentrations and smaller mussel sizes generally coinciding.

The calculations for dock turnover times do not take filtration by other filter feeders such as ascidians into account. Randlov & Riisgård (1979) gave filtration rates of 0.43 and 0.29 mls $^{-1}$ (1.5 and 1.0 l hr^{-1}) for average sized individuals of *Ciona intestinalis* and *Ascidiella aspersa*

respectively, rates comparable to those of average sized *Mytilus edulis* individuals. Ascidians, barnacles and other suspension feeders are commonly found amongst the dominant mussel cover, so in this respect calculated dock volume filtration rates are under estimated.

6.4.2.2 Observed and potential beneficial effects of biological filtration on water quality

The loss of a control dock with low mussel populations due to natural settlement soon after the start of the project was unfortunate. However, a great deal of information can be gained by examining the changes in water quality which took place after introduction or natural settlement of mussels. The remarkable improvements in water clarity and reductions in chlorophyll *a* seen in the Albert and Graving Docks are attributed to the control of phytoplankton by biological filtration which shows a large increase over the same time period. This decrease in phytoplankton biomass was achieved despite generally longer hours of bright sunshine in summer 1989 and 1990 compared to 1988 (see fig 2.2 c). In the Queens Dock increased filtration by mussels failed to control phytoplankton populations. The rate of turnover of dock water by mussels is slower in Queens than in Albert Dock, and to a lesser degree Graving Dock. In addition several features are present in the Queens Dock which could reduce the impact of such filtration. The Queens Dock does not benefit from the well distributed mussel populations present in the Graving Dock, and to a lesser extent the Albert Dock. A greater surface area of mussel bed to dock volume ratio allows better filtration of the whole water body. Additionally, the Graving Dock benefits from artificial mixing which increases flow of water over the mussels and so reduces refiltration of water. Evidence that increased mixing aids biological filtration was seen in August 1989, when an increase in mixer speed was accompanied by a sharp increase in water clarity, although this may also have been due to direct effects of mixing on phytoplankton production. The shallow nature of the Queens Dock which allows rapid rise in temperature, ready nutrient supply from the sediments and the concentration of phytoplankton in a shallow, well lit water column, will further favour the development of high algal biomass.

The improved hypolimnetic oxygen concentrations seen in the Albert Dock over the summers 1988 to 1990 may be due, at least in part to increased biological filtration. The occurrence of thermal stratification was also less frequent from June to August in 1989 and 1990 and this will certainly account for a proportion of the improvements in oxygen concentrations. In 1988, however, low oxygen levels occurred without the presence of thermal stratification and in May 1990 oxygen depletion was not observed even during a 2 - 3 week period of thermal stratification. This reduced tendency for the development of anoxic conditions may be due to the filtering action of the mussels controlling phytoplankton biomass and hence oxygen depletion during die-off and subsequent microbial breakdown. It is also possible that thermal stratification itself was reduced as a consequence of decreased phytoplankton biomass. It has been shown that increased absorption of solar radiation due to high phytoplankton biomass enhances the rate of heating at the ocean surface and reduces the distribution of heat through the water column (Sathendranath *et al*, 1991). Hence the decrease in phytoplankton biomass seen in 1989 and 1990 compared to 1988 may have resulted in less thermal stratification. This in itself has important implications for water quality and is another potential beneficial effect of biological filtration (see general discussion). A decrease in phytoplankton standing crop could also have been caused by reduced nutrient concentrations or increased zooplankton grazing. These factors can be eliminated because nutrient concentrations were similar in all three years and zooplankton populations fell dramatically in the two later years.

It is possible that a mussel filter may be capable of reducing levels of undesirable bacteria introduced with replenishment water. *Mytilus edulis* has been shown to be capable of retaining and breaking down several species of bacteria, including *Esherichia coli* (Birbeck & McHenery, 1982). Such effects would only be of benefit if pathogenic micro-organisms such as *Salmonella* were also removed. No detailed studies of the effect of mussel filtration on bacterial populations have been carried out in the South Docks. Tests for faecal coliform bacteria are carried out by Altwell Ltd and levels have been found to be generally within EC guidelines for bathing waters, despite the high levels in the river outside.

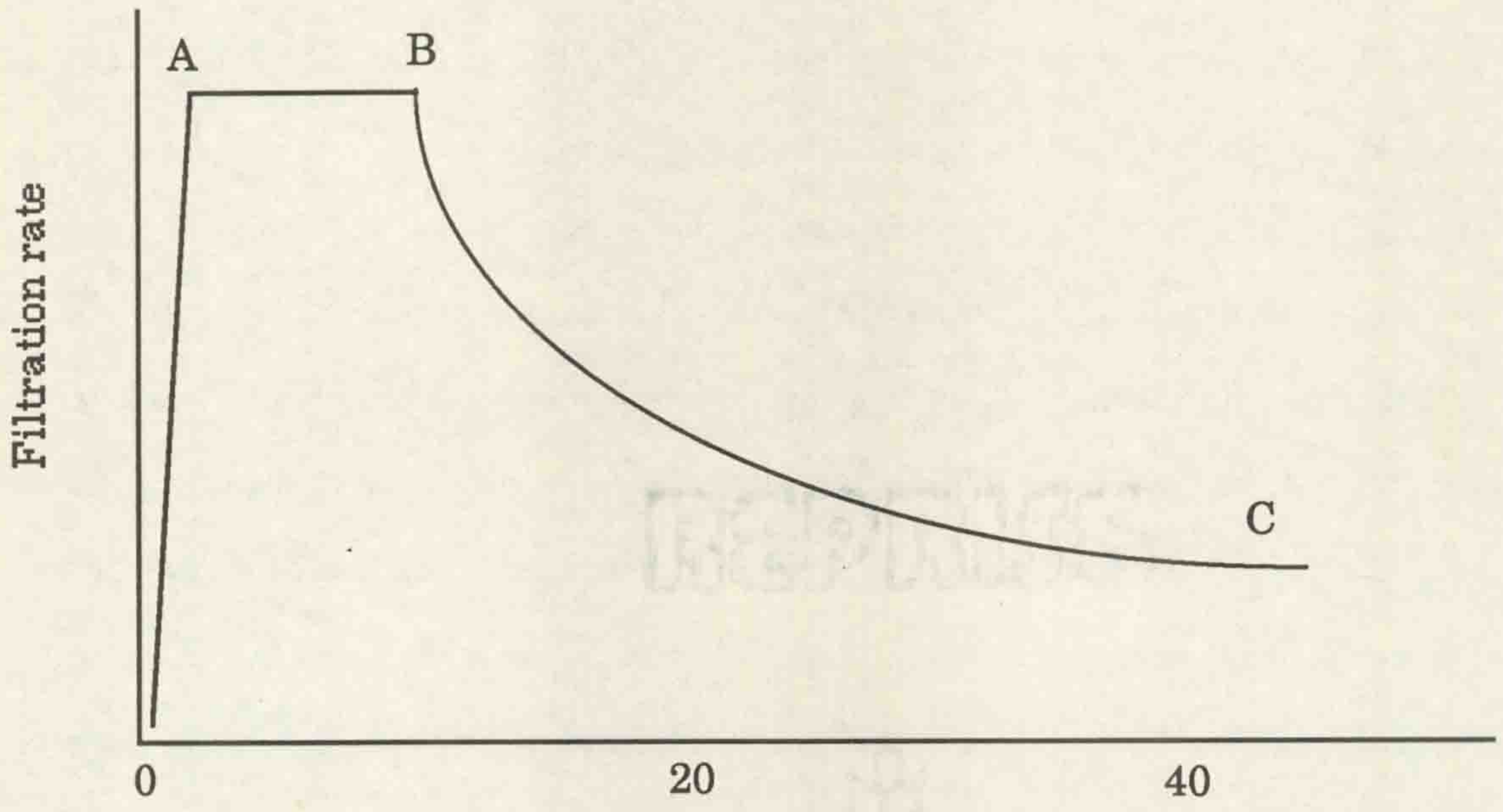


Fig 6.11 Concept of the interrelationships existing between filtration rate and food concentration (12 ºC). Modified from Winter (1978).

6.4.2.3 The mechanisms of control of phytoplankton by the mussel population

The mechanisms by which mussels may control phytoplankton biomass need some consideration here. In most predator-prey relationships, and in particular zooplankton/ phytoplankton interactions, an oscillating pattern of prey increase followed by increased predator numbers, leading to subsequent decline of prey is seen. Given the great difference in life span of mussels and phytoplankton it is clear that mussels cannot control phytoplankton by an increase in biomass in the short term. Additionally, in zooplankton-regulated systems a rapid rise in phytoplankton biomass in the spring is possible due to low numbers of existing zooplankton. A lag phase occurs in which zooplankton may be both temperature and food limited, during which phytoplankton biomass can increase relatively unchecked.

In contrast, in a mussel-controlled system predator biomass is similar throughout the year. If predation rates in such a system were also constant, it might be expected that phytoplankton biomass would be greatly suppressed at all but the seasonally highest rates of primary production, however, this is not the case. In the Albert and Graving Docks, when densities of mussels were high, phytoplankton biomass was still able to increase in early spring. It was controlled, however, later in the year when light levels were suitable for higher rates of primary production. This problem can be partly explained by examining the relationship between phytoplankton cell concentration and filtering rate of mussels (Fig 6.11 after Winter 1978). At very low cell concentrations, such as occur at the end of the winter, filtration rates are also low. Colder water temperatures at this time will also reduce filtration rates. Consequently, phytoplankton can take advantage of increasing daylength in February and March and biomass will increase. In spring, filtration rates will increase with rising phytoplankton densities until a maximum rate is reached (point A Fig. 6.11). Mussels continue to filter water at the same rate between points A and B on the curve and will remove all phytoplankton cells from this water volume (effective retention is down to 3µm , Jørgenson

Table 6.7 Comparison of water volume filtration (turnover) time and calculated particle concentration halving time of bivalve populations, between several coastal marine embayments. * - partial control of phytoplankton, Δ - no control of phytoplankton, all other areas - control of phytoplankton biomass reported. Turnover times for South Docks are calculated from Smaal *et al* 1986.

Source	Area	Water volume filtration time. (days)	Particle conc. Halving time (days)
Cloern (1982)	S. San Fransisco Bay	1.2 - 1.8	0.8 -1.3
Cohen et al (1984)	Potomac River	3 - 4	2.1 - 2.8
Nichols (1985)	N. San Francisco Bay	1	0.7
*Smaal et al (1986)	Oosterschelde	4 - 5	2.8 - 3.5
Loo & Rosenberg (1989)	Laholm Bay	3	2.1
Hily (1991)	Bay of Brest	3 - 4	2.1 - 2.8
-	Albert Dock	1.5	1.0
-	Graving Dock	3.9	2.7
Δ -	Queens Dock	4.8	3.3

1975), so that more cells will be removed at higher concentrations despite a constant filtration rate. For this part of the curve the degree of predation is controlled by the prey concentration rather than changes in numbers of predators or rates of filtration. A maximum rate of ingestion is achieved at point B (Fig 6.11) above which filtration rates decline with increasing phytoplankton. At point C pseudofaeces production begins and filtered algae which are surplus to mussel requirements are lost to the sediments. Up to concentration B (Fig 6.11) increasing algal biomass will result in greater mussel removal rates, above B filtration rates decline and phytoplankton biomass could rise more quickly. Increases in temperature will have a stimulatory effect on filtration rates up to between 15°C and 20°C (Winter 1978). These are typical water temperatures in the South Docks between June and September.

If the rate of cell removal is equal to the rate of reproduction then phytoplankton biomass will remain constant. Just as the rate of phytoplankton cell division can be expressed as the doubling time, the rate of removal of plankton cells can be described by the halving time. The halving time is related to the turnover time of a dock volume of water through the mussels but is not simply the time taken to filter half the volume of water. The mussels in such an enclosed system are continually removing particles and diluting the suspension with filtrate. If the filtering rate remains constant then the rate at which particles are removed will progressively decline, that is, as the concentration decreases so do the decrements, successive decrements bearing a fixed ratio to the existing concentration. Decrements are removed at infinitely small intervals so that concentration is a continuous function as expressed by the curve e^{-x} (Coughlan 1969). This assumes homogeneity of the dock water at all times, 100% retention rate of algal cells and no replication of the cells. This relationship has been used for the indirect measurement of filtration rates from the removal of particles in suspension.

One general equation for this is given by Coughlan (1969) as:

$$m = \frac{M}{n} . \log_e \frac{C_o}{C_t}$$

Where m is the filtering rate, M is the volume of suspension, C_0 is the initial concentration and C_t the concentration after time t, assuming no settlement of algae, n is the number of animals per test vessel.

From this relationship particle halving time in a water body, by a population of filter feeders (t) can be calculated assuming that the filtration rate of the population (m) is known. C_0 / C_t would now = 2 and n may be ignored. The time taken for a halving of phytoplankton cell concentrations, assuming no reproduction can be expressed as:

$$t = \frac{M}{m} . \log_e 2$$

Given that M / m is the dock volume filtration time (T):

$$t = T . 0.693$$

Obviously this is an oversimplification: assumptions such as homogeneity of dock water and absence of sinking of algae are unlikely to be true; in open bays or estuaries horizontal transport of algae would occur. However, in waterbodies with a long hydrodynamic residence times a rough estimate of the filtration turnover rates needed for control of phytoplankton can be made from maximum algal division rates. In areas where filter feeders are reported to control phytoplankton the range of water volume turnover times and particle concentration halving times is remarkably small (Table 6.7). Halving times calculated from these figures range from 0.7 to 2.8 days compared to average measured and maximum expected phytoplankton doubling times of 1 to 1.5 days and 0.4 to 0.7 days respectively, for nutrient rich waters in the Southern Hemisphere (Parsons *et al* 1977). These figures are surprisingly well matched given the inaccuracies inherent in estimations of filtration rates. They suggest that matching numbers of filter feeders to algal growth rates in such a way could be used as a 'rule of thumb' when attempting biological control at other sites. Thus it would appear from

studies in other areas and in the South Docks that a water volume filtration time of less than about 4 days, or a particle concentration halving time of less than 3 days, is required for adequate control of phytoplankton by benthic filter feeders (Table 6.7).

6.4.2.4 The effects of mussel cultivation on nutrient cycling

The heterotrophic activity of mussels increases the rate of transfer of nutrients from the particulate organic to dissolved phase (Dame *et al* 1989). This supply of inorganic nitrogen and phosphorus may form an appreciable part of the supply available to pelagic production (Kautsky & Wallentinus 1980, Kautsky & Evans 1987). Therefore, while dense populations of filter feeders may keep phytoplankton biomass low, they also have the potential to stimulate primary production by increasing dissolved nutrient levels (Kautsky & Wallentinus 1980, Kaspar *et al* 1985, Smaal *et al* 1986, Asmus & Asmus 1991, Hily 1991). The supply of nutrients from regeneration by filter feeders is at its greatest at exactly the time of year when nutrient concentrations might otherwise be limiting (Asmus & Asmus 1991). One effect of increasing the biomass of filter feeders in the South Docks may therefore be to increase phytoplankton production, despite reducing the problem of nuisance phytoplankton biomass. Such nutrient regeneration is likely to have a greater stimulatory effect on phytoplankton if a large proportion of the particulate food was from non-planktonic sources (e.g. from organically rich sediments). In the South Docks the amount of non-planktonic seston is low, so the effect of filter feeders is simply to return nutrients more quickly to the planktonic phase. Mussel filtration may have the effect of smoothing out phytoplankton blooms by reducing the rate of biomass build up and by moderating nutrient related crashes of populations.

The cultivation of mussels may bring about a net removal of nitrogen from the system. This is achieved by a high rate of bacterial denitrification by the rope community, burial of organic matter by rapid deposition and, in farmed cultivation, a removal of nitrogen locked up in

mussel tissue at harvesting (Kaspar, 1985). Denitrification requires an anoxic/oxic boundary in sediments to proceed (Vries & Hopstaken 1984). Such interfaces would be provided on mussel ropes which are covered in organically rich detritus (Kaspar 1985). As N is the most limiting nutrient in the South Docks, this would be an important step towards reducing eutrophication. The encouragement of natural settlement rather than introductions of biomass from outside is obviously desirable from the perspective of nutrient reductions. Continual placement of new settlement ropes and removal of older mussels would reduce the total amount of nitrogen and phosphorus in the dock ecosystem, whereas mortality of introduced mussels would result in additions of nutrients. Additionally, the replacement of older ropes would keep the size of mussels small. Large numbers of small mussels are preferable to the same biomass of larger mussels because smaller mussels have a higher weight specific filtration rate (Winter 1978, Officer *et al* 1982).

The actual effects of mussel populations on dissolved nutrients in the South Docks cannot be determined due to the influence of other factors such as increased nutrient inputs from the estuary in later years.

6.4.2.5 Potential Problems And Deleterious Effects

The encouragement of very dense populations of mussels in an enclosed water body such as the South Docks presents several possible problems.

Artificial mixing is required to enable mussel cultivation in docks prone to severe oxygen depletion. In the Graving Dock mussel ropes tested before artificial mixing showed 100% mortality below 2 m depth within 14 days, due to low oxygen levels. After mixing no such mortality was observed on the 5 m long ropes. Once a filter feeding population of sufficient size is established, net improvements in oxygen concentrations may be observed and the requirements for mixing may be reduced.

Other authors have reported negative effects of dense benthic mussel populations on the dissolved oxygen saturation in bottom waters (Jørgensen 1980, Lopez *et al* 1984, Kaspar *et al* 1985. The oxygen requirements of mussel beds are high. Oxygen consumption by mussel beds has been found to be as much as 10 times greater than by surrounding benthic communities (Jorgensen, 1980). The large amount of organic detritus settling from mussel ropes may exceed the amount that can be utilized by infaunal organisms, resulting in a build up of anoxic sediments with a high rate of hydrogen sulphide production (Dahlbäck & Gunnarson, 1981). Mussel cultivation could therefore exaggerate the problems of low oxygen saturation at sensitive periods, resulting in mortality of benthic fauna.

Intensive rope mussel culture will locally affect the benthic ecosystem. In coastal ecosystems the diversity of infauna beneath ropes is likely to decrease, with a switch to opportunistic polychaete species, due to the build up of organic-rich anoxic sediments (Tenore *et al* 1982, Mattson & Linden 1983, Lopez *et al* 1984, Kaspar *et al* 1985). However, increased macro-epifaunal production (providing extra food for fish), both on and under mussel ropes, is likely, compared to surrounding areas (Tenore *et al*, 1982, Romero *et al*, 1982; Lopez *et al*, 1984; Kaspar, 1985). When ropes were first installed in the Graving Dock such effects were not a problem since there was no benthic fauna. It is likely that a low diversity of benthic fauna would be typical of any eutrophic area in which biological control by mussels might be considered.

Large amounts of ammonia are released from mussel ropes (Kaspar, 1985). Ammonia is toxic under certain conditions (Bower & Bidwell, 1978) and mussel cultivation could elevate concentrations to dangerous levels in docks where concentrations are already high. Again, combining biological filtration with artificial mixing may reduce this problem, maintaining the levels of dissolved oxygen required for nitrification of ammonia to nitrate.

A lowering of N/P ratios is possible in enclosed areas with dense mussel cultivation due to increased denitrification. It has been suggested that this shift, combined with the high productivity of such areas, may provide ideal conditions for the formation of dinoflagellate blooms (Rodhouse & Roden 1987, Folke & Kautsky 1989), but no such links have yet been identified. In the South Docks the trend in those docks with higher mussel densities was away from phytoplankton dominated by dinoflagellates.

The presence of dinoflagellates may reduce the efficiency of control by filter feeders. A bloom of the dinoflagellate *Gyrodinium aureolum* was shown to cause reduced filtration rates and marked gut damage in *Mytilus edulis* (Widdows *et al* 1979b) The lowest filtration rates measured for Albert Dock mussels were recorded during a *Prorocentrum minimum* / *Prorocentrum micans* bloom (after transfer to Queens Dock).

It is possible that the dramatic decline in zooplankton populations seen after the arrival of mussels was caused by filtration effects. If this is the case then mussels may be removing larval stages brought in with river water and may have a negative effect on the recruitment of new species in the docks.

6.4.3 Future management of water quality in the South Docks

On completion of the studies of water quality and experimental management techniques, recommendations for the future control of water quality were made to the Merseyside Development Corporation. These recommendations are at varying stages of implementation.

With the exception of the Graving Dock, the Albert Dock is the only other Dock likely to benefit from the installation of a water mixer to improve oxygen concentrations, as all other docks are shallow and likely to have sufficient natural mixing. The maintenence of high dissolved oxygen levels is of particular importance in the Albert Dock due to its high public profile and the desire to ensure survival of the resident mussel population.

'Off the shelf' mixers are expensive and the most commonly used mixer in docks, the Helixor, could not be used in the Albert Dock as it would not allow a deep enough draught for boats. I therefore designed a low-cost, perforated pipe-compressed air type mixer for the Albert Dock. The requirements of such a system were calculated according to Davies (1980) using the following design criteria:

1) Mixer capability to overturn density discontinuity resulting from 4°C temperature drop between 3 m and 4 m depth with no wind mixing (approximation to worst recorded case, June 1988).

2) Required destratification time - 1 day.

3) Perforated pipe length of 100 m to be used.

The calculated mixer requirements were as follows:

1) A compressor supplying oil free air at a rate of 7.56 ls-1 and a pressure (at the compressor) of 2.8 bar. If this pressure is exceeded efficiency is lost as maximum water entrainment is achieved at a low overpressure.

2) 115 m of medium density polyethylene pipe, internal bore 40.8mm (pressure grading to 12 bar) having 100 m perforated section with 1mm holes drilled every 0.3 m. Pipe sealed at submerged end.

3) Anchor weights of total in water weight of 200 kg required to sink pipe.

Operation of this mixer began in August 1991 (see Plate 6.1). Regular measurements of dissolved oxygen are now taken by MDC staff. Mixing is only carried out at night unless oxygen concentrations are low, this reduces noise levels and visual impact when the area is busy and will also help to keep water temperatures low.

There is no appropriate rapid solution to the problem of dinoflagellate blooms in Queens Dock. The mortality of mussels added over the sediments shows that direct additions of mussels to the dock bottom cannot be used. This mortality is probably due to an anoxic micro-

layer above the sediments as mussels are capable of growing on soft sediments elsewhere (Verwey 1952, Theisen 1968, Smaal *et al* 1986). The original settlement structures (scallop shells and tyres) proved to be poor collectors of filter feeders. Tyre rossettes with a mesh covering (see Plate 6.2) are likely to be better collectors but the large area of mesh needed to provide a sufficient dock volume filtration rate (estimated at around 2000 m^2 to double the existing filtration rate) makes this an expensive option (approx. £1000 for mesh plus costs for construction and installation). The installation of mesh and tyre units are presently under investigation. This may be undertaken on a large scale in future. The best chance of biological control in Queens Dock, however, must lie with the natural establishment of a healthy benthic filter feeding infauna. The shallow, well mixed nature of this dock would enhance the ability of such an assemblage to control phytoplankton. Burrowing bivalves have been found in low densities in the Albert Dock in most recent samples and it is possible that numbers will increase throughout the South Docks with time.

The input of water from the Mersey cannot be avoided. The addition of water in smaller, more frequent amounts, as carried out in recent years, should help to reduce the penetration of water and bacteria through the docks and restrict the area of sediment build up. I advised against the use of Mersey tunnel water due to the lower salinity and higher dissolved nutrient levels than Mersey River water. I considered that the higher dissolved nutrient levels would stimulate phytoplankton blooms and that lowered salinity might cause mortality of beneficial fauna and could result in conditions suitable for growth of blue-green algae.

In summary, the water quality problems in the South Docks are essentially those of eutrophication. Inputs of nutrient rich water from the Mersey cannot be prevented, so tactical solutions must be considered. Artificial mixing was shown to reduce thermal stratification and increase oxygen concentrations. Mixing may also inhibit in the development of dense growths of dinoflagellates. Biological filtration by mussels appeared to dramatically reduce phytoplankton blooms, although an adequate experimental control was lacking. Most

Plate 6.1 Albert Dock (February 1992), showing perforated pipe air lift mixer in operation.

Plate 6.2 Experimental tyre and mesh reef unit.

of the problems that could result from intensive mussel cultivation are considered either to be irrelevant in eutrophic systems, or to be ameliorated by the use of artificial mixing. The applicability of biological filtration techniques to other systems and needs for further research are discussed in chapter 7.

CHAPTER 7

GENERAL DISCUSSION

7.1 SHORTFALLS IN METHODOLOGY AND NEEDS FOR FURTHER WORK

In retrospect there are several areas of this research project which could have been done in a better way. One major problem, as mentioned in chapter 1, was that it was impossible to plan for the full 33 month term from the outset. Had this been possible more attention would have been given to the benthos of the dock walls and sediments in the initial stages. It would also have been preferable to have started earlier in the first year in order to have a complete spring - summer cycle for comparison with later years. These restrictions were due to the nature of funding and could not be avoided.

The lack of a control dock that was suitably matched to the experimental Graving Dock in terms of size, shape, depth and exposure was another problem to which no solution was available. The use of parallel controls would have been much preferable to the comparison of results from before and after treatments. The Albert Dock provided the best approximation to a matched control initially, as it is reasonably deep and had very low mussels populations at the start of the project. The heavy settlement of mussels in autumn 1988 prevented the use of the Albert Dock as a control for comparison with the mussel introductions into the Graving Dock.

The study of plant nutrients in the South Docks was based only on measurements of the dissolved inorganic/reactive phase. A better understanding of the relative importance of nutrient recycling within the docks, compared to external supply, would have been gained if nutrients of the dissolved organic, particulate and unreactive phases had also been taken into account, preferably in both dock water and estuarine 'top-up' water. Constraints of time prevented this.

In the Albert and Queens Docks plankton sampling could only be carried out from pontoons close to the edge of the dock as no boat was available. Sampling across the docks would have shown the degree of within dock patchiness. This would have been of interest, particularly in relation to the distribution of mussels. No formal studies of the mega-zooplankton or fish, and only a very limited study of the sediment benthos, was carried out. In order to build a picture of the ecosystem as a whole it would have been of interest to study these components in more detail.

Despite the above problems with sampling methods it is considered that the results are adequate to show the major long term changes and spatial variation in the physico-chemical and biological parameters under investigation.

This work has highlighted several areas which require further research:

1) The initial 3 - 4 years of colonisation after re-introduction of water was not recorded in the South Docks. Observations from this period are vital for the understanding of successional patterns in high salinity docks. The imminent dredging and redevelopment of Wallasey Dock, on the Wirral, provides an ideal opportunity to study the initial colonisation period.

2) A more detailed study of the sediment fauna of the South Docks would allow a better assessment of the importance of this component of the dock ecosystem, especially in terms of possible interactions with water quality.

3) The sharp decline in zooplankton numbers seen after mussel settlement suggests some direct or indirect interaction. An analysis of the nature of such effects and the possible implications for benthic recruitment needs to be addressed.

4) It is possible that mussels may remove bacteria from the water column. This would only reduce the potential health hazard if pathogenic micro-organisms were removed in addition to indicator species. Detailed study of the ability of mussels to remove microbial pathogens, and their ultimate fate, is required to assess any advantage provided by mussel populations in this respect.

5) Further work is required to establish the relative importance of introduced versus internal recycling of nutrients.

7.2 THE VALUE OF RESTORED DOCKS

Consideration of the value of restored dock complexes can be split into those aspects which were planned at the development stage and those which have been produced as a spin off. The main aim of redevelopment schemes is to revitalise a derelict or under-used area in the inner city and to stimulate future economic activity with a combination of viable business, housing and amenity projects. The South Docks complex now incorporates luxury housing, office space, retail, cultural and recreational useage. The project has certainly been successful in terms of general appearance of the area and as a tourist attraction. Whether the project will be a financial success in the long term is more uncertain. Rents for retail outlets are expensive and units are frequently vacant. Several proposals for major projects, for example a national aquarium, have failed to attract investors. As at other redeveloped docks there has been criticism that the luxury nature of the housing projects have excluded local buyers. Apart from the marina, water-based recreational facilities are generally under-used, despite heavily subsidised equipment hire and instruction charges. It is possible that this is due to lack of public confidence in the water quality.

All development projects require a high standard of water quality, which, as already discussed, is not always satisfied. The most potentially serious problem is that of risk to public health. The waters of the South Docks generally conform to E.C. bathing water guideline levels for coliform bacteria (Altwell 1988, 1989, McAleese 1988), however, occasional

localised high counts are obtained. This, combined with the inadequacies of the use of coliform indicator species for assessing health risks, is a cause of concern for the Merseyside Development Corporation. At Bristol Docks a gastroenteritis outbreak occurred in participants of a snorkel event, despite the fact that the water conformed to E.C. guidelines for bathing waters (Evans *et al* 1983). No complaints of illness amongst watersports users have been received at the South Docks. Because of worries about public health the MDC only encourages secondary contact watersports such as windsurfing and canoeing, although other activities such as SCUBA diving are carried out at the risk of the individual.

The potential for conservation and educational use is an incidental advantage of redeveloped docks, which may be exploited more in the future. The conservation value lies not only in the direct provision of aquatic habitat, but also in the provision of areas for housing, recreation, pleasure-boat mooring and business in the heart of cities, thus reducing pressure on greenfield sites.

It has been suggested that redeveloped docks can provide an 'oasis' for aquatic life in urbanised areas (Hendry 1988a, Hawkins *et al* in press b). In the South Docks a reasonably diverse flora and fauna has developed. The docks are frequently used by seabirds and occasionally ducks, and a group of cormorants regularly feed in the dock waters. Unfortunately, the ability of docks to develop a diverse flora and fauna is restricted by the low habitat diversity and, in many cases poor water quality. Water quality management, carried out for purely commercial reasons can improve the conditions for aquatic life and increase species diversity (e.g. Russell *et al* 1983). Docks are constructed with vertical, featureless walls and a flat sediment bottom. In flat, sandy coastal areas artificial reefs, made of various materials, have been successfully used to increase three dimensional habitat diversity, usually in order to increase fish catches (Bohnsack & Sutherland 1985, Downing *et al* 1985, Martin & Kelly 1985). The use of artificial reefs in the South Docks would almost certainly improve its worth as a habitat for marine life, but this must be undertaken with care. The materials of which such reefs are constructed must be of low toxicity as any substances released from introduced

materials would quickly accumulate in the docks. The toxicity of materials may be difficult to determine, for example tyres are known to contain benthiazoles, persistent organic chemicals for which little information on toxicity is available, although they are generally thought to be harmless (Spies *et al* 1987). Care would also need to be taken that artificial reefs did not interfere with the passage of boats or future dredging activities, and that no safety hazard would be presented to watersports activities.

The physical conditions in docks may be suitable for colonisation by lagoonal species, excluding those typical of fringing communities. Several species found in the South Docks during this research project have been listed as specialist lagoonal species (Barnes 1988b). Barnes (1991) has suggested that threatened lagoonal species could be introduced to specially constructed man made lagoons. Redeveloped docks may provide a suitable habitat for such introductions, thus reducing the need to scour out new lagoon sites.

Redeveloped docks are potentially a valuable resource for educational and research institutions. Educational projects are compatible with most existing redevelopments and research projects are generally welcomed by the development agencies. The urban location of docks means that they are accessible to a large number of groups. Providing that water quality is adequate to support a variety of aquatic organisms, docks may be used, for example, for school projects, collection of demonstration material, research projects and, perhaps most importantly, for environmental education of the general public. Non-biological subjects may also be taught with the help of a visit to a dock complex, such as local, commercial or shipping history.

The South Docks have been used for a number of BSc, and short MSc research projects, and are now used for teaching of university field studies in marine biology. In summer 1989 a free exhibition on the marine life of the docks was staged in an empty shop unit in the Albert Dock ('Living Waters'). The exhibition included aquaria containing dock biota, an underwater video, binocular microscopes for the examination of live material and display boards. This exhibition attracted in the order of 30,000 people over three weeks and was repeated on a

larger scale by the National Museums And Galleries On Merseyside in summers 1990 and 1991. This exhibition ('Dockwatch'), was staged in the Maritime Museum at the South Docks, and attracted over 70,000 people each year. Exhibitions of this kind can bring the conservation message home to a broad spectrum of the population - particularly if entrance is free! The live displays and local aspect will appeal to many people who would not normally take an interest in such matters. Such displays were particularly appreciated by children and school groups.

7.3 ECOSYSTEM FUNCTIONING

At the start of this research project, in early May 1988, the South Docks ecosystem was dominated by the plankton. Dense phytoplankton and zooplankton populations were found in all docks, while benthic flora and fauna were relatively sparse. From this time onwards the benthos has played an ever increasing part in the functioning of the ecosystem. This was initially seen by an increase in benthic filter feeders, living on the dock walls, which could utilize the abundant planktonic food supply. The diversity of macroalgae on the dock walls has also increased over the last three years. The improved water clarity observed in some docks over this time period has allowed greater depth penetration of such macrophytes. Such a progression from a planktonic to benthic dominated food web is more likely to occur in systems of higher salinity where powerful benthic filter feeders are more common. Oligohaline systems such as Preston Dock are unlikely to develop in this direction due to a paucity of suitable species (Conlan 1989). In shallow, freshwater lakes an increase in benthic primary production is often associated with control of phytoplankton blooms by appropriate management (Moss 1990). Such a progression is unlikely to occur due to control by natural benthic filter feeders, because, again, suitable species are very few.

A simplified food web for the South Docks is illustrated in Fig. 7.1, in which only the main non-detrital pathways are shown. No attempt at quantification of biomass has been made for any components of this food web. It is clear, however, that the predominance of filter feeding and planktonic heterotrophs and sparsity of benthic herbivores illustrates that the

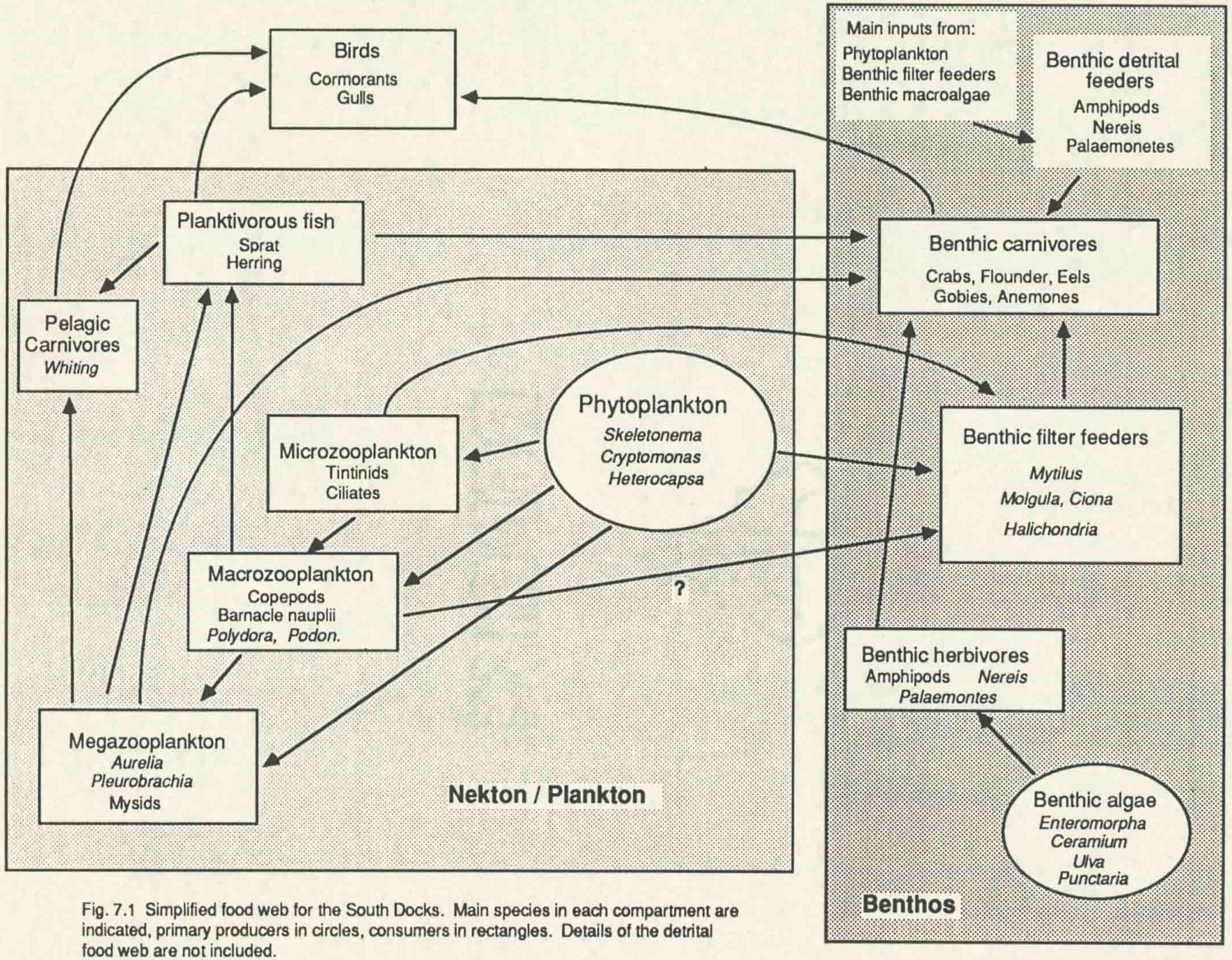


Fig. 7.1 Simplified food web for the South Docks. Main species in each compartment are indicated, primary producers in circles, consumers in rectangles. Details of the detrital food web are not included.

ecosystem is still largely driven by planktonic primary production. Obligate benthic herbivores are extremely rare, most consumers of macrophytes currently present in the South Docks are omnivorous. This is possibly due to the short time that macrophytes have been present in the South Docks and colonisation by benthic herbivores may occur with time. *Littorina littorea* and *Littorina saxatilis* are the most likely benthic herbivores to survive in the South Docks as the low salinities will prevent survival of most marine rocky shore benthic herbivores.

At present the wall benthos is much more dense than the sediment dwelling fauna, probably due to low oxygen concentrations in the sediments. Signs of increased density and diversity of sediment dwelling fauna have been seen recently (see chapter 5) and it seems likely that this trend will continue. The wall fauna is dominated by filter feeding species, but a rich associated detrital feeding fauna is also found (see Appendix Tables XV to XII), presumably feeding on organically rich faeces and pseudofaeces produced by the filter feeding organsims.

The South Docks shows a predominance of species common in brackish waters, despite the relatively high salinities. It is possible that the salinity is just below the limits of tolerance for some common coastal species. It is known for example that *Asterias rubens* is present in large numbers in some of the North Docks, and were present in Sandon, in salinities only 1 - 2 °/oo higher than the South Docks. *A. rubens* introduced with mussel ropes in the Graving Dock were observed to survive for up to 12 months, but, despite an abundant food supply, eventually disappeared all together. Starfish are capable of greatly reducing mussel populations (Paine 1966), so the inability of *Asteias rubens* to survive in the South Docks may have a major impact on the wall community, allowing continued domination by *Mytilus edulis*.

A very schematic representation of the timing and sequence of colonisation of the walls in the South Docks is shown in Fig. 7.2. This diagram is based on observations from Albert Dock. Other docks show the same sequence of colonisation, although the percentage cover of each group of species and timing relative to return of water vary somewhat. No observations have

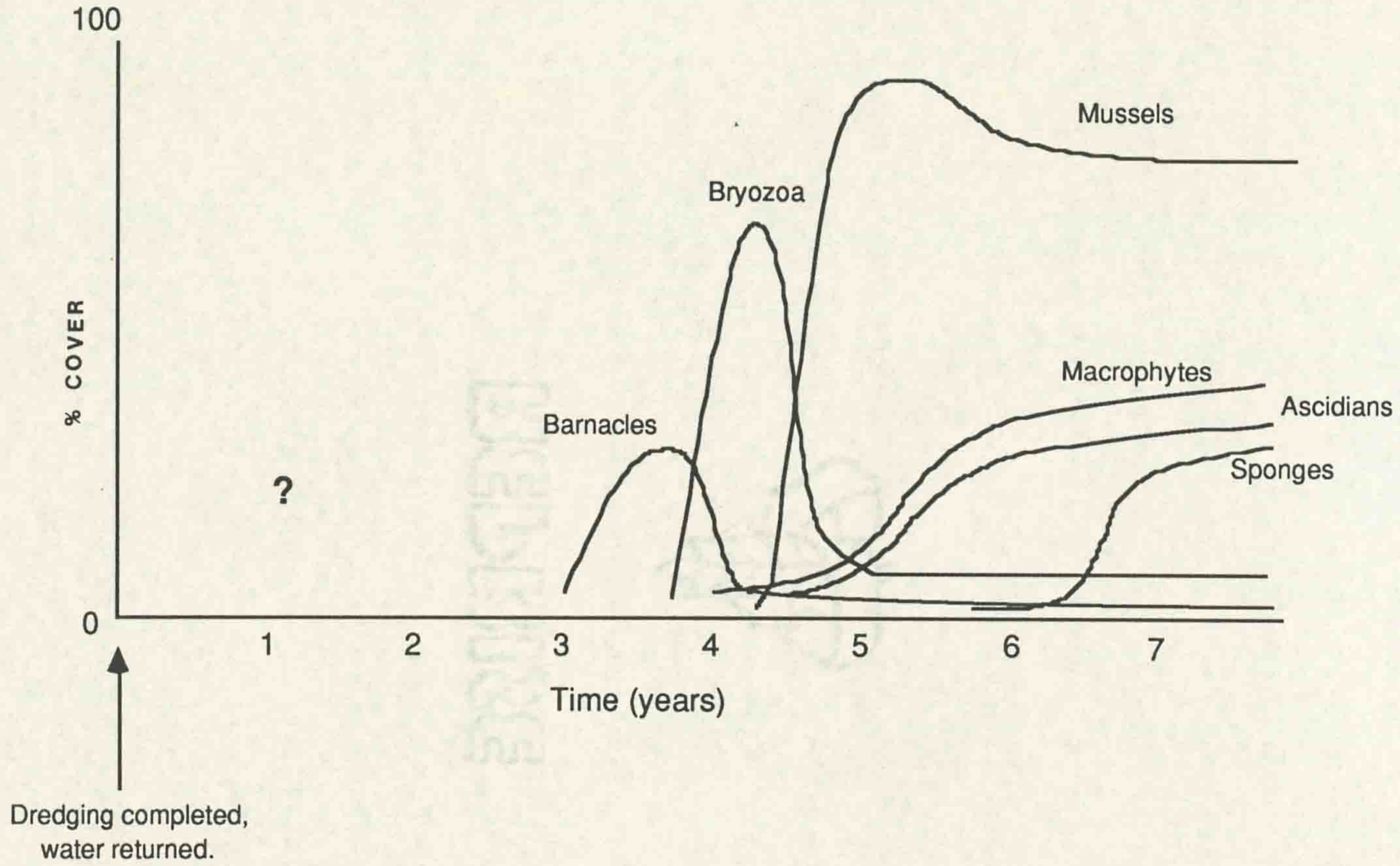


Fig. 7.2 Schemetic diagram of colonisation of Albert Dock walls after dredging and refilling with water.

been made of the wall biota in the first three to four years after completion of dredging. The progression after this time is thought to be the domination of walls by barnacles then bryozoans, then possibly a brief settlement of hydroids followed by the arrival of mussels, with a gradual increase in macrophyte cover and numbers of ascidians, and finally the appearance of *Halichondria panicea*. Some variation in benthic community structure is likely to occur in the South Docks even if a climax community is reached. Year to year variation in larval recruitment, for example, due to food limitation or larval supply, may lead to years of higher phytoplankton levels and subsequent improved recruitment. The mechanisms of this succession and comparison with other systems is discussed previously in chapter 5.

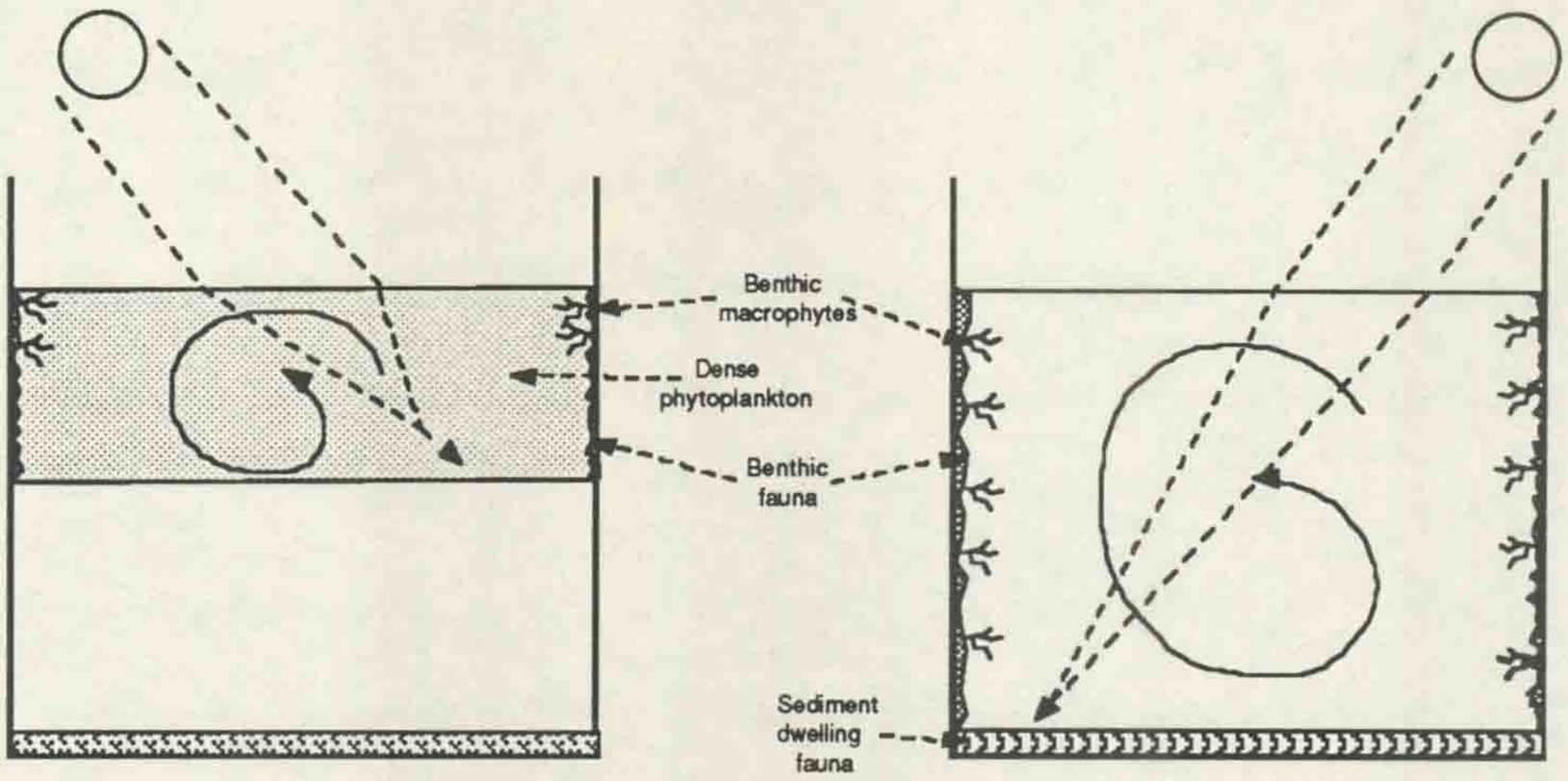
It has been recognised for some time that many natural communities have multiple stable states (Holling 1973, May 1977, Sutherland 1974). The reaction of freshwater lakes to eutrophication provides one of the best examples of this, a phytoplankton rich turbid state being associated with eutrophication and clear water state associated with low nutrient loading. In many cases the two opposing states are described simply as typical communities existing in either nutrient rich or nutrient poor conditions (e.g. Holling 1973, Krebs 1978). The resilience of freshwater systems to return to clear water conditions when nutrient loading is reduced is often stressed. With the recent attention to water quality control by biomanipulation, some authors have described positive feedback mechanisms which can maintain the alternative communities, even at the same nutrient concentrations (Hosper & Jagtman 1990, Moss 1990, Scheffer 1989, 1990).

Most biomanipulation methods in freshwaters rely on the encouragement of high densities of large bodied herbivorous zooplankters, either by removal of fish predators or by the provision of macrophyte refuges from predation (Moss 1990). Identification of positive feedback mechanisms have therefore tended to concentrate on the fish / zooplankton / macrophyte interactions (see Moss 1990, Scheffer 1990 for overviews). The turbid water condition in shallow lakes is typified by high phytoplankton biomass, very limited distribution

of macrophytes, high densities of planktivorous and benthiviorous fish and low numbers of large bodied zooplankton. The opposite situations are found in the clear water state where the high water clarity allows the growth of macrophytes which provide a refuge from predation for large bodied zooplankton, therefore maintaining grazing pressure and keeping phytoplankton densities low and water clarity high. Other buffering mechanisms may also take place, such as reduced benthic bioturbation by fish.

The sudden switch from turbid to clear water conditions in the South Docks provides further evidence for the existence of the two stable states, or, to use the terminology of Holling (1973), the two domains of attraction. When the main components of the system which represent each state are considered along with their positive (enforcing) or negative (repelling) interactions it is apparent that a large number of possible interactions tend to enforce the existence of the alternative states and to resist transferral from one to the other. These are summarized in (Fig. 7.3) which includes some buffering mechanisms which are not considered by authors detailing the fish - zooplankton - macrophyte based control of phytoplankton biomass (e.g.Hosper & Jagtman 1990, Moss 1990, Scheffer 1989, 1990). Most interactions within the high phytoplankton / low filter feeder density system and within the low phytoplankton / high filter feeder density states show enforcement of other factors typical of that state. Most interactions between the two states are negative, resisting transfer from one state to the other. This schematic diagram (Fig 7.3) only includes those factors which are thought to have a significant impact on the existence of either state in the South Docks. The effects of zooplankton and fish populations, which are important in freshwater systems were not considered to be an important influence on the changes seen in the South Docks and have been omitted for simplicity.

The main enforcing interaction within the turbid state is the reduction of hypolimnetic dissolved oxygen concentrations, which prevents colonisation by benthic filter feeders. Dense phytoplankton populations may cause oxygen depletion during the decay of blooms and by increasing absorbtion of solar radiation in surface layers, thus enhancing thermal

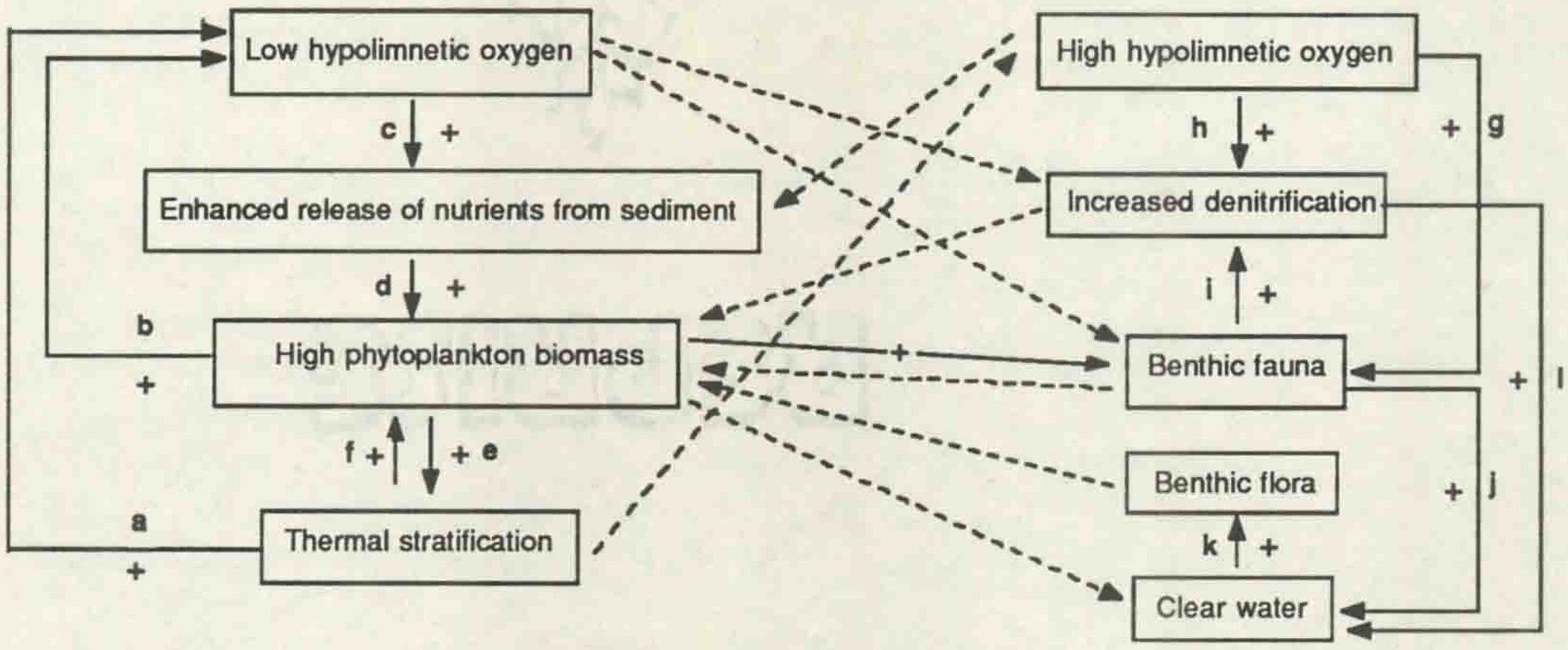


TURBID WATER STATE

Biomanipulation

Arificial mixing

CLEAR WATER STATE



Mechanisms of effect

a) Reduced atmospheric oxygenation (Pastorok et al 1981)
b) Oxygen depletion during bloom die off (Reynolds & Walsby 1975)
c) Phosphate release in anoxic sediments (Mortimer 1971)
d) Sediment supply of nutrients for phytoplankton growth (Stefan & Hanson 1980)
e) Increased absorbtion of radiation in surface layers (Sathendranath et al 1991)
f) Concentration of phytoplankton in high light zone (Pingree et al 1975)
g) Survival of benthic fauna (Fast 1973)
h) Oxic / anoxic sediment boundry (Webb 1981)
i) Increased anoxic / oxic boundary (Kaspar et al 1985)
j) Benthic control of phytoplankton (Officer et al 1982)
k) No light limitation of macrophyte growth (Moss 1990)
l) Nitrogen limitatation of phytoplankton growth (Ryther & Dunstan 1971).

Fig. 7.3 Model of possible alternative stable states in the South Docks and associated feedback mechanisms.

stratification (Sathendranath *et al* 1991). In the clear water state the filter feeders reduce algal blooms and benthic oxygen concentrations remain high, ensuring the continued survival of benthic fauna. The effects of high phytoplankton or high filter feeder densities on nutrient dynamics and macrophyte populations are also included, but are considered to be of lesser importance. Water quality management by mixing or biomanipulation aims to push the system from one state to another in the hope that feedback mechanisms will ensure the continuation of the clear water state once this is achieved. In the presence of a larval supply of filter feeders mixing alone may be sufficient action, as natural recruitment of filter feeders should occur when conditions are made favourable. Biomanipulation by the addition of a filter feeding population will speed the progression, or enable it to happen where a suitable larval supply is unreliable or non-existent.

Scheffer (1990) states that biomanipulation can only cause a sustainable shift to a clear water state if nutrient levels have been reduced to a certain level. At higher concentrations the phytoplankton will eventually escape zooplankton control and reversal to a turbid state will follow. This could happen in benthic controlled systems if a phytoplankton outbreak resulted in oxygen depletion and mortality of fauna. Observations from natural systems show that in shallow marine systems control of phytoplankton by benthic suspension feeders is very resistant to high nutrient levels when mixing of water is good ensuring oxygenation of the water (Officer *et al* 1982, Hily 1991). Artificial mixing would help to ensure survival of filter feeding benthos at all depths in enclosed systems, so that occasional phytoplankton blooms or prolonged hot weather would not cause reversal to the turbid water state. Mixing would also increase the rate of supply of phytoplankton to the filter feeders.

As described in chapter 6 the longevity of mussels when compared to phytoplankton will enhance their ability to control phytoplankton from the initial stages of a bloom and avoids the instability of zooplankton controlled systems due to rapid changes in populations. The lag period between increases in phytoplankton and zooplankton populations in spring, which allows temporal escape of the phytoplankton production, does not apply to benthic control by

filter feeders such as mussels. Much of the instability of zooplankton control, however is due to predation effects. In the South Docks at present the number of predators of mussels is low, being restricted mainly to *Carcinus maenas* which can only take small mussels. If a more effective mussel predator such as *Asterias rubens* appeared this could have devastating effects, as would the appearance of disease or parasites. *Asterias ruben* did not survive in the South Docks after inadvertent introduction and it is possible that the salinity of the water is too low. The stability of the benthic filter feeding control is therefore likely to be greatly enhanced by using assemblages of filter feeding species, e.g mussels, ascidians and sponges. A natural progression to this state seems to be occurring in the South Docks. A 'whole ecosystem' approach to management, taking into account all possible interactions and aiming for a broad spectrum of control species is likely to result in greatest stability of the clear water state. The use of artificial reefs for collection of filter feeders may also provide refuges for zooplankton from fish predation. Interestingly this approach increases the conservation value of the docks.

7.4 THE APPLICABILITY OF MANAGEMENT TECHNIQUES TO OTHER SYSTEMS

Artificial mixing is a tried and tested method of reducing thermal stratification and improving dissolved oxygen concentrations in fresh waters. This use of mixing in most marine systems is precluded by the sheer size of the water bodies concerned. However, its use in small marine enclosures such as docks or marine lakes is a feasible option if good water quality is required.

For the use of biofiltration techniques several features are demanded of the water body. Flow through of water must be slow enough to allow time for filtration of the water, and mixing must be sufficient to bring all the water column into the proximity of the animals. Water quality must be good enough to ensure survival of the animals; problems of low dissolved

oxygen concentrations may be eliminated by the use of artificial mixing. Other environmental factors must also allow survival: for example the range of temperatures, pH and presence of toxic substances must all be taken into account. A suitable substrate for attachment or burial of the chosen species must be present. A high three dimensional heterogeneity will allow the maximum diversity of species and biomass of animals, thus increasing stability and filter feeding pressure.

Attention must also be given to the possible deleterious effects of introductions on the existing flora and fauna. As biological control would only be considered in severely degraded environments this would not normally be a problem.

If introductions of filter feeders, rather than natural settlement is to take place careful consideration of appropriate species is needed. There is a great need for further research in this area. In high salinity systems there is a large variety of filter feeding organisms available. The feasibility of using sediment dwelling species such as *Macoma balthica* or *Cardium glaucum* is one area that would merit further attention. *Cardium glaucum* is a species typical of saline lagoons and has a high tolerance of low oxygen saturations (Barnes 1980). In oligohaline and freshwater systems the number of filter feeding species available is much reduced. In areas of very low salinity, species diversity as a whole is very low, reaching a minimum at 5 - 7 °/oo (Wetzel 1975). The polycheate *Ficopamatus enigmaticus* has been shown to be a powerful filter feeder (Davies *et al* 1989), but this is a reef building species which would preclude its use in docks to be used for watersports or boating. Research is needed to identify suitable species. Baltic strains of mussels grow in salinities as low as 4°/oo and are one possibility, but great care is needed when introducing foreign strains as these animals may introduce parasites to commercial operations, or may become a pest species if escaping the intended environment.

In fresh waters the number of benthic filter feeding species is limited and freshwater species tend to have lower filtration rates than their marine counterparts. For example *Dreissena polymorpha* is ecologically similar to *Mytilus edulis* in many ways, however, its filtration rates are much lower (Reeders 1989, H.D. Jones pers. comm.). *Dreissena* is also a major pest in inland waterways in Europe and North America, causing blockage of pipes and channels, hence movement of this animal is severely restricted. The possiblity of introducing *Dreissena* to isolated waters such as Salford Quays is being considered by workers at Manchester University, but some form of sterilisation to prevent reproduction, such as induction of triploidy, is likely to be required. Given the vast numbers of animals needed to provide sufficient filtering power the use of *Dreissena* without the possibility of *in situ* reproduction to boost numbers seems an unlikely option.

The use of both biofiltration and artificial mixing could possibly clash with intended water use. Structures associated with these techniques, such as buoys, nets and 'Helixor' tubes may present an obstacle to shipping. Subsurface ropes and nets may present a hazard to watersports users if inappropriately positioned. Artificial reefs may cause problems if repeated dredging operations will be required. Floating buoys associated with suspended filter feeder culture might be aesthetically unacceptable, although use of existing structures such as pontoons is often an available option. Air-lift mixers such as 'Helixor' and perforated pipe systems require an on-shore compressor to supply air. This may cause problems due to high noise levels, and effective sound proofing is usually required.

The introduction and running costs of water quality controls are often considerable. However, this is likely to be a fraction of the overall cost of the redevelopment of disused docks. Consideration of the possible effects of factors such as dredging depth, bottom material, three dimensional heterogeneity, water supply and degree of isolation on water quality should be made at the construction stage, as this can reduce costs in the long-term.

In conclusion, this study of the South Docks has provided an insight into the ecology, water quality and possibility for water quality improvements, in high salinity docks. It has wider implications to other enclosed water bodies and in some cases to the coastal environment as a whole. The work in particular demonstrates the resilience of aquatic ecosystems, and the speed of recovery which can take place when a pollution or stress factor is reduced.

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Appendix Table I Orthophosphate-P mg/l (n = 3)

	Grav. surf.		Grav. 5m		Grav. 9m		Albert surf.		Albert 5m		Queens surf.	
	Mean	S.E	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.
14-Jun	0.07		0.32		0.44		0.15		0.24		0.21	
28-Jun	0.16	0.02	0.41		0.51	0.01						
12-Jul	0.56	0.00	0.60	0.00	0.68	0.00	0.48	0.01	0.49	0.01	0.52	0.00
27-Jul	0.55	0.00	0.55	0.00	0.57	0.01						
9-Aug	0.61		0.68		0.76	0.00	0.49		0.64		0.58	0.01
23-Aug	0.65	0.07	0.74	0.03	0.77	0.01	0.63	0.02	0.66	0.01	0.70	0.02
20-Sep	0.70	0.02	0.73	0.00	0.75	0.02	0.67	0.00	0.65	0.00	0.66	0.00
17-Oct	0.49	0.01	0.50	0.00	0.47	0.01	0.36	0.00	0.37	0.00	0.37	0.01
1989												
19-Jan	0.43	0.00	0.42	0.01	0.43	0.00	0.40	0.01	0.39	0.00	0.38	0.00
22-Feb	0.24	0.00	0.25	0.00	0.25	0.00	0.23	0.00	0.22		0.15	
15-Mar	0.29	0.00	0.26	0.02	0.29	0.02	0.24	0.01	0.25	0.00	0.24	0.01
17-Apr	0.16	0.00	0.18	0.00	0.20	0.01	0.18	0.00	0.18	0.00	0.19	0.00
17-May	0.39	0.00	0.40	0.01	0.42	0.01	0.37	0.01	0.36	0.01	0.36	0.01
14-Jun	0.50	0.01	0.58	0.05	0.60	0.05	0.47	0.01	0.60	0.01	0.61	0.01
28-Jun	0.34	0.01	0.35	0.00	0.40	0.00	0.27	0.01	0.28	0.00	0.31	0.00
12-Jul	0.47	0.02	0.49	0.00	0.61	0.01	0.48	0.00	0.47	0.01	0.43	0.00
26-Jul	0.21	0.00	0.22	0.00	0.36	0.00	0.22	0.00	0.28	0.03	0.13	0.01
9-Aug	0.53	0.01	0.58	0.00	0.86	0.02	0.59	0.02	0.60	0.04	0.54	0.01
30-Aug	0.36	0.02	0.37	0.00	0.39		0.38	0.01	0.39		0.40	0.00
19-Sep	0.77	0.02	0.78	0.00	0.65	0.00	0.60	0.01	0.57	0.00	0.61	0.00
13-Oct	0.81	0.02	0.84	0.01	0.84	0.00	0.64	0.01	0.63	0.00	0.63	0.00
15-Nov	0.45	0.00	0.45	0.00	0.42	0.01	0.36	0.01	0.37	0.01	0.37	0.01
13-Dec	0.42	0.06	0.41	0.04	0.34	0.04	0.19	0.01	0.13	0.02	0.27	0.02
1990												
23-Jan	0.33	0.01	0.38	0.00			0.34	0.01			0.35	0.00
14-Feb	0.19	0.01	0.20	0.00	0.24	0.00	0.12	0.00	0.15	0.01	0.21	0.00
13-Mar	0.37	0.01					0.34	0.00			0.34	0.00
6-Apr	0.16		0.19		0.15		0.16		0.15		0.06	
10-May	0.01		0.01		0.01	0.00	0.01	0.00	0.01	0.00	0.01	0.00
7-Jun	0.18	0.03	0.24	0.00	0.24	0.01	0.24	0.01	0.19	0.01	0.18	0.02
18-Jul	0.24	0.01	0.26	0.00	0.30	0.01	0.19	0.00	0.17	0.01	0.19	0.00
9-Aug	0.62	0.03	0.66	0.09	0.70	0.02	0.44	0.00	0.44	0.02	0.56	0.00
12-Sep	0.67	0.01	0.59	0.04	0.60	0.02	0.44	0.04	0.51	0.03	0.55	0.01

Appendix Table II Nitrate + nitrite - N mg/l (n = 3)

Date 1988	Grav. surf.		Grav. 5m		Grav. 9m		Albert surf.		Albert 5m		Queens surf	
	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.
26-May												
14-Jun												
28-Jun	0.01		0.07		0.04							
12-Jul	0.01	0.00	0.02	0.01	0.01	0.00	0.04	0.01	0.01	0.00	0.02	0.00
27-Jul	0.08	0.01	0.07	0.01	0.10	0.01						
9-Aug	0.19		0.17		0.13		0.13		0.08			
23-Aug												
20-Sep	0.35	0.00	0.35	0.00	0.34	0.01	0.06	0.00	0.06	0.01	0.11	0.00
17-Oct	0.37	0.01	0.38	0.00	0.34	0.00	0.24	0.01	0.24	0.00	0.15	0.00
14-Nov	0.52	0.02	0.49	0.00	0.50	0.00	0.43	0.00	0.42	0.00	0.45	0.00
12-Jun												
19-Jan	0.98		0.96		0.95		0.96		0.95		1.00	
22-Feb	0.78	0.01	0.77	0.00	0.76	0.01	0.73	0.01	0.70	0.01	0.58	0.01
15-Mar	0.49	0.01	0.60	0.04	0.63		0.24	0.02	0.18	0.01	0.01	0.00
17-Apr	0.11	0.01	0.09	0.00	0.12	0.01	0.05	0.00	0.16	0.04	0.04	0.00
17-May	0.01	0.00	0.01	0.00	0.16	0.06	0.08	0.05	0.07	0.03	0.05	0.02
14-Jun	0.04	0.01	0.03	0.01	0.05	0.00	0.02	0.01	0.06	0.01	0.03	0.02
28-Jun	0.01	0.00	0.01	0.00	0.02	0.00	0.08	0.01	0.07	0.00	0.02	0.01
12-Jul	0.01	0.00	0.01	0.00	0.01	0.00	0.01	0.00	0.08	0.01	0.01	0.00
26-Jul	0.01	0.00	0.01	0.00	0.01	0.00	0.01	0.00	0.01	0.00	0.01	0.00
9-Aug	0.01	0.00	0.02	0.01	0.02	0.00	0.06	0.02	0.10		0.04	0.01
30-Aug	0.01	0.00	0.01	0.00	0.01	0.00	0.01	0.00	0.01	0.00	0.01	0.00
19-Sep	0.49	0.03	0.52	0.00	0.40	0.01	0.22	0.04	0.28	0.00	0.31	0.00
13-Oct	0.53	0.00	0.53	0.00	0.54	0.00	0.33	0.00	0.33	0.00	0.41	0.00
15-Nov	0.65	0.00	0.65	0.00	0.57	0.01	0.73	0.01	0.64	0.02	0.71	0.04
13-Dec	0.76	0.02	0.77	0.01	0.75	0.01	0.70	0.01	0.68	0.01	0.72	0.01
13-Jun												
23-Jan	1.11	0.03	0.99	0.01			1.09	0.00			1.15	0.01
14-Feb	1.07	0.02	1.08	0.00	0.84	0.01	1.19	0.01	1.18	0.00	1.26	0.01
13-Mar	1.22		1.28				1.28				1.22	
6-Apr	0.80	0.02	0.84	0.00	0.76	0.02	0.85	0.01	0.90	0.01	0.55	0.01
10-May	0.42	0.01	0.44	0.01	0.50	0.00	0.22	0.01	0.49	0.01	0.09	0.01
7-Jun	0.16	0.01	0.16	0.01	0.21	0.00	0.26	0.00	0.27	0.00	0.16	0.01
18-Jul	0.19	0.01	0.20	0.00	0.17	0.00	0.04		0.18	0.00	0.01	0.00
9-Aug	0.09	0.00	0.10	0.00	0.05	0.01	0.04	0.00	0.05	0.00	0.01	0.00
12-Sep	0.28	0.01	0.29	0.00	0.22	0.00	0.22	0.00	0.20	0.00	0.20	0.01

0.01 = At or below limit of detection

Appendix Table III Ammonia - N mg/l (n = 3)

Date 1988	Grav. surf.		Grav. 5m		Grav. 9m		Albert surf.		Albert 5m		Queens surf	
	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.
14-Jun	0.22	0.02	0.78	0.21	0.91	0.05	0.22		0.29		0.28	
28-Jun	0.10	0.00	0.48	0.08	0.61							
12-Jul	0.29	0.01	0.42	0.00	0.64	0.02	0.22	0.00	0.16	0.01	0.93	0.02
27-Jul	0.19	0.01	0.23	0.00	0.27	0.01						
23-Aug	0.26	0.00	0.29	0.03	0.33	0.01	0.35	0.01	0.33	0.01	0.41	0.01
20-Sep	0.25	0.00	0.25	0.00	0.29	0.01	0.35	0.00	0.48	0.00	0.47	0.00
17-Oct	0.11	0.00	0.12	0.00	0.13	0.00	0.10	0.00	0.11	0.00	0.13	0.01
1989												
19-Jan	0.22	0.01	0.23	0.00	0.24	0.01	0.20	0.00	0.19	0.00	0.21	0.00
22-Feb	0.11	0.00	0.10	0.00	0.10	0.00	0.05	0.00	0.09	0.00	0.09	0.00
15-Mar	0.04	0.00	0.04	0.00	0.08	0.00	0.06	0.00	0.06	0.00	0.06	0.00
17-Apr	0.03	0.00	0.04	0.00	0.15	0.00	0.06	0.00	0.06	0.01	0.06	0.01
17-May	0.09	0.00	0.10	0.00	0.35	0.02	0.11	0.01	0.10	0.00	0.13	0.00
14-Jun					0.34	0.01						
28-Jun	0.10	0.00	0.13	0.02	0.41	0.01	0.12	0.00	0.25	0.00	0.27	0.00
12-Jul	0.10	0.00	0.09	0.00	0.41	0.02	0.09	0.00	0.16	0.06	0.21	0.04
26-Jul	0.08	0.00	0.08	0.00	0.61	0.01	0.08	0.01	0.09	0.00	0.19	0.05
9-Aug	0.10	0.00	0.10	0.00	0.73	0.04	0.08	0.00	0.09	0.01	0.22	0.02
30-Aug	0.09	0.01	0.10	0.02	0.11		0.08	0.00	0.08	0.00	0.08	
19-Sep	0.38	0.00	0.39	0.00	0.58	0.00	0.13	0.00	0.26	0.00	0.34	0.02
13-Oct	0.45	0.00	0.47	0.00	0.47	0.00	0.37	0.01	0.49	0.00	0.50	0.00
15-Nov	0.48	0.00	0.47	0.01	0.71	0.01	0.57	0.02	0.52	0.00	0.57	0.01
13-Dec	0.42	0.00	0.42	0.00	0.43	0.00	0.45	0.00	0.45	0.00	0.45	0.00
1990												
23-Jan	0.23	0.00	0.38	0.03			0.32	0.00	0.33	0.01		
14-Feb	0.18	0.00	0.18	0.00	0.38	0.01	0.14	0.00	0.16	0.00	0.17	0.00
13-Mar	0.08	0.01					0.08	0.01	0.07	0.00		
6-Apr	0.09	0.01	0.08	0.00	0.76	0.02	0.10	0.00	0.08	0.00	0.07	0.00
10-May	0.07	0.00	0.05	0.00	0.22	0.01	0.07	0.00	0.06	0.01	0.07	0.01
7-Jun	0.10	0.01	0.08	0.00	0.13	0.00	0.02	0.00	0.07	0.01	0.06	0.00
18-Jul	0.04	0.00	0.04	0.01	0.28	0.01	0.01	0.00	0.01	0.00	0.05	0.00
9-Aug	0.22	0.00	0.23	0.01	0.38	0.01	0.02	0.01	0.09	0.01	0.07	0.00
12-Sep	0.22	0.00	0.20	0.01	0.26	0.01	0.05	0.01	0.21	0.01	0.22	0.00

0.01 = At or below limit of detection

Appendix Table IV Reactive Silicate - Si (mg/l) (n = 3)

1990	Graving surface Mean	S.E.	Graving 5m Mean	S.E.
23-Jan	0.84		0.99	0.01
14-Feb	0.73	0.00	0.73	
13-Mar	0.61	0.01		
6-Apr	0.17		0.18	0.00
10-May	0.07	0.01	0.08	
7-Jun				0.00
18-Jul	0.07	0.00	0.06	0.01
9-Aug	0.37	0.00	0.40	0.01
12-Sep	0.49		0.44	

1990	Graving 9m Mean	S.E.	Queens surface. Mean	S.E.
23-Jan	0.99			0.01
14-Feb	0.67	0.00	0.70	
13-Mar	0.58	0.02		
6-Apr	0.16		0.14	0.01
10-May	0.07	0.01	0.08	
7-Jun				0.00
18-Jul	0.01	0.00	0.01	0.00
9-Aug	0.02	0.01	0.01	0.01
12-Sep	0.24	0.01	0.27	

1990	Albert surface Mean	S.E.	Albert 5m Mean	S.E.
23-Jan			0.98	0.00
14-Feb	0.90	0.00	0.75	0.02
13-Mar			0.48	
6-Apr	0.19		0.16	0.01
10-May	0.08	0.01	0.05	
7-Jun				0.00
18-Jul	0.22	0.01	0.01	0.01
9-Aug	0.70	0.03	0.53	0.02
12-Sep	0.35	0.00	0.11	

0.01 = At or below limit of detection

Appendix Table V Chlorophyll a µg/l (n = 3)

DATE	Grav. surf.		Grav. 5m		Grav. 9m		Albert surf.		Albert 5m		Queens surf.	
	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.
14/6/88	10.7	6.8	1.8	0.3	2.3	0.9	3.0		8.5		3.0	
21/6/88	11.0	2.6	6.4	0.7	2.3	1.5	17.0		0.0		30.3	
27/6/88	11.7	3.5	41.0	4.0	6.7	0.3						
12/7/88	71.0	6.9	8.3	0.9	6.3	0.7	33.7	2.7	15.0	0.0	13.0	0.7
27/7/88	35.0	1.0	22.0	1.0	11.3	0.7						
9/8/88	45.7	0.7	10.3	0.3	0.7	0.7	12.0	0.0	1.5	0.8	13.8	0.8
23/8/88	23.3	0.9	8.0	0.6	2.3	0.3	10.3	2.7	3.7	0.7	7.2	2.6
20/9/88	3.3	1.7	2.0	0.0	2.0	0.0	3.0	1.0	3.0	1.0	5.4	0.8
4/10/88	30.0	9.0	21.7	1.8	23.3	1.2						
17/10/88	7.1		7.0		6.0		12.0		7.0		16.6	
14/11/88	1.0	0.0	1.0	0.0	1.0	0.0	1.7	0.3	1.0	0.0	5.4	0.8
19/12/88	0.0		0.0		0.0	0.0	0.0		0.0			
19/1/89	0.0		0.0		0.0	0.0	0.0		0.0		2.8	0.6
22/2/89	5.9		5.7	0.3	1.3	0.3	1.0	0.6	1.2	0.7	13.4	0.8
15/3/89	22.3	3.3	24.0	0.0	30.0	3.0	66.3	5.4	53.7	6.3	79.1	3.2
17/4/89	8.7	1.1	12.7	2.6	2.7	0.7	5.1	1.7	2.0	1.4	19.0	0.7
17/5/89	8.3	1.2	5.3	2.4	1.3	1.3	2.8	0.4	0.8	0.8	0.8	0.8
14/6/89	4.7	1.9	5.0	0.6	4.7	0.3	3.6	0.0	7.5	0.8	2.8	1.4
28/6/89	7.3	0.3	7.3	2.0	2.3	1.9	3.6	0.7	4.4	1.4	23.3	1.4
12/7/89	5.0	0.0	4.0	1.0	2.0		3.2	1.4	6.3	1.7	28.1	3.9
26/7/89	10.0	1.5	11.3	0.3	0.7	0.3	6.7	0.4	5.5	2.4	33.7	4.8
9/8/89	13.3	0.7	4.7	1.3	12.0	1.0	17.4	2.0	5.1	1.0	36.7	3.0
23/8/89	8.0	2.5	2.7	1.3	6.7	0.7	7.5	0.4	18.9	1.8	28.8	0.8
19/9/89	7.1		2.0	0.0	0.7	0.7	4.7	1.4	3.9	2.1	41.1	1.6
11/10/89	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
15/11/89	0.0	0.0	0.0		0.0		0.0	0.0	0.0	0.0	0.0	0.0
13/12/89	0.0	0.0	0.0		0.0		0.0	0.0	0.0	0.0	0.0	0.0
23/1/90	5.0		1.0				0.8		0.0		0.0	
14/2/90	1.0		0.0		0.0		1.2		0.0			
12/3/90	3.0						1.2		0.0		4.7	
6/4/90	3.3	0.7	8.3	2.4	4.7	0.7	2.7		16.4		8.6	0.4
10/5/90	2.3	0.9	3.3	2.0	11.0	4.4	2.6	1.1	3.0	1.1	13.6	3.3
7/6/90	4.3	0.3	3.3	2.0	4.0		1.9	1.8	3.6	1.2	19.6	3.6
18/7/90	7.3	1.3	11.3	5.8	3.7	0.7	10.1	1.0	19.0	1.1	33.0	6.6
9/8/90	0.7	0.3	0.7	0.7	3.0		1.2	0.7	4.4	1.4	10.9	1.7
12/9/90	0.6		0.8	0.8	2.3	1.4	1.2	1.2	1.5	0.8	12.9	0.7

Appendix Table VI

GRAVING DOCK	MEAN PHYTOPLANKTON NUMBERS PER CM3							
SURFACE								
DATE	9-Jun 1988	21-Jun	28-Jun	12-Jul	9-Aug	23-Aug	20-Sep	17-Oct
CRYPTOPHYCEAE								
Cryptomonas sp.	0	0	11	0	0	0	0	506
EULGENOPHYCEAE								
Grouped Euglenoids	2063	1355	570	8098	12452	12606	117	21
DINOFLAGELLATES (Dinophyceae)								
Amphidinium sp.	0	0	0	0	0	0	0	0
Gymnodinium sp	45	3195	12795	7137	0	0	0	2
Gyrodinium spirale	0	0	0	0	0	0	20	18
Micracanthodinium claytonii	0	0	0	0	15	7	0	36
Prorocentrum minimum	0	0	0	32	0	86	0	2
Protoperidinium bipes	0	0	0	0	0	0	0	0
Scripsiella	0	0	11	0	0	14	<1	0
Total Dinoflagellates	45	3195	12806	7169	15	107	20	58
DIATOMS (Bacillariophyceae)								
Chaetoceros	0	0	0	0	0	0	0	4
Leptocylindricus .	0	0	0	0	0	0	0	1
Melosira	0	0	0	0	0	0	0	0
Naviculoid Diatoms	173	414	0	42	25	43	8	1
Rhizosolenia setigera	0	0	0	0	0	0	<1	1
Thalassiosira sp.	0	0	0	0	0	0	12	75
Unidentified centric diatoms (10 u diameter)	1883	0	0	53	164	66	28	10
Total Diatoms	2056	414	0	95	189	109	48	92
TOTAL PHYTO >10u	4164	4964	13387	15362	12656	12822	185	678
Small Flagellates (4 to 10µ diameter) Phaeocystis clumps present	2504	3325	84	1667	10389	1916	347	278
PLANKTONIC PROTOZOA (ciliates,oligotrichs,tintinnids)	0	471	106	0	113	90	22	8

Appendix Table VI cont.

GRAVING SURFACE	MEAN PHYTOPLANKTON NUMBERS PER CM3							
DATE	14-Nov	19-Dec	19-Jan 1989	22-Feb	15-Mar	17-Apr	17-May	15-Jun
CRYPTOPHYCEAE								
Cryptomonas sp.	8	95	36	673	363	481	244	429
EULGENOPHYCEAE								
Grouped Euglenoids	<1	10	3	18	40	74	3	0
DINOFLAGELLATES (Dinophyceae)								
Amphidinium sp.	0	<1	0	10	0	0	8	0
Gonyaulax tamarensis	0	0	0	0	0	0	132	2
Gymnodinium sp	0	0	0	12	2	21	9	0
Gyrodinium spirale	0	1	0	2	0	0	1	0
Heterocapsa triquetra	0	0	0	0	0	665	0	0
Katodinium sp.	0	0	0	0	0	6	0	0
Microanthodinium claytonii	0	0	0	0	17	17	0	4
Prorocentrum minimum	0	0	0	0	0	9	34	0
Protoperidinium bipes	3	0	0	21	0	0	0	0
Scripsiella	0	0	0	17	70	0	9	0
Total Dinoflagellates	3	1	0	62	89	718	159	6
DIATOMS (Bacillariophyceae)								
Chaetocerous sp.	0	0	0	<1	63	0	0	0
Leptocylindrus danicus	0	0	0	36	680	0	1	10
Lithodesmium	0	0	0	0	0	0	1	10
Melosira	0	4	0	0	0	0	0	0
Naviculoid Diatoms	<1	2	0	6	6	0	1	0
Rhizosolenia setigera	0	<1	0	0	4	8	0	0
Skeletonema costatum	0	0	0	5	209	0	0	0
Thalassiosira sp.	2	0	0	45	247	0	14	11
Unidentified centric diatoms (10 u diameter)	2	0	<1	181	<1	0	0	8
Total Diatoms	4	6	0	268	1209	8	17	39
TOTAL PHYTO >10u	15	17	36	1021	1701	1281	457	474
Small monads & flagellates (4 to 10µ diameter)	16	17	7	458	25339 mph	2974 mph	119	163
Phaeocystis clumps present				+				
PLANKTONIC PROTOZOA (ciliates,oligotrichs,tintinnids)	1	4	5	7	32	30	118	181

GRAVING SURFACE	MEAN PHYTOPLANKTON NUMBERS PER CM3							
DATE	28-Jun 1989	12-Jul	26-Jul	9-Aug	23-Aug	19-Sep	11-Oct	15-Nov
CRYPTOPHYCEAE								
Cryptomonas sp.	2232	290	1536	703	119	33	19	32
EULGENOPHYCEAE								
Grouped Euglenoids	13	6	0	6	74	0	0	1
DINOFLAGELLATES (Dinophyceae)								
unid. dinoflagellate	0	0	6	101	0	0	0	0
Amphidinium sp.	3	0	0	0	0	0	0	0
Gonyaulax tamarensis	0	0	0	0	0	0	0	0
Gymnodinium sp	3	13	19	57	0	0	0	0
Gyrodinium spirale	8	0	10	0	0	0	0	0
Heterocapsa triquetra	0	0	0	0	0	0	0	0
Katodinium sp.	5	0	0	0	0	0	0	0
Micracanthodinium claytonii	3	0	3	0	3	0	0	0
Oxyrrhis marina	0	0	0	3	0	0	0	0
Prorocentrum minimum	61	1560	76	405	0	3	6	0
Protoperidinium bipes	0	0	0	0	0	0	0	0
Scripsiella	5	0	0	0	0	0	0	0
Total Dinoflagellates	88	1573	111	566	3	3	6	0
DIATOMS (Bacillariophyceae)								
Chaetoceros sp.	5	13	6	0	0	9	0	0
Leptocylindrus danicus	10	0	0	51	0	9	0	0
Lithodesmium	0	0	0	0	0	0	4	0
Melosira	3	0	0	0	0	3	0	0
Naviculoid Diatoms	0	90	6	32	5	4	4	0
Rhizosolenia setigera	13	0	0	0	18	4	0	0
Skeletonema costatum	0	0	0	127	71	71	218	0
Striatella	0	0	0	0	10	0	0	0
Thalassiosira sp.	15	184	10	545	0	0	0	0
Unidentified centric diatoms	0	0	16	0	0	0	0	0
(10 u diameter)	0							
Total Diatoms	43	287	38	755	104	100	226	0
TOTAL PHYTO >10u	2376	2156	1685	2030	556	136	251	33
Small Flagellates (4 to 10µ diameter) Phaeocystis clumps present	426	109	329	1121	226	75	39	10
PLANKTONIC PROTOZOA (ciliates,oligotrichs,tintinnids)	426	22	76	32	23	3	0	1

Appendix Table VI cont.

GRAVING SURFACE	MEAN PHYTOPLANKTON NUMBERS PER CM3							
DATE	13-Dec	23-Jan 1990	14-Feb	13-Mar	29-Mar	6-Apr	24-Apr	3-May
CRYPTOPHYCEAE								
Cryptomonas sp.	9	405	496	189	175	125	154	285
EULGENOPHYCEAE								
Grouped Euglenoids	1	0	2	1	264	11	15	19
DINOFLAGELLATES (Dinophyceae)								
Amphidinium sp.	0	0	0	0	0	0	0	0
Gymnodinium sp	0	0	1	0	2	3	0	0
Gyrodinium spirale	1	0	0	0	2	0	0	0
Heterocapsa triquetra	0	0	0	0	0	1	0	19
Katodinium sp.	0	0	0	0	0	0	0	19
Micracanthodinium claytonii	0	0	0	0	0	4	0	0
Oxyrrhis marina	0	0	0	1	0	0	3	19
Prorocentrum minimum	0	0	1	1	0	0	0	0
Protoperidinium bipes	0	0	0	1	0	0	0	0
Scripsiella	0	0	0	0	2	0	0	0
Total Dinoflagellates	1	0	2	3	6	7	3	57
DIATOMS (Bacillariophyceae)								
Ceratulina	0	0	0	0	0	0	0	19
Chaetocerous sp.	0	0	0	15	0	0	0	0
Leptocylindrus danicus	0	0	0	0	0	0	0	0
Melosira	4	0	0	0	0	0	0	0
Naviculoid Diatoms	1	0	0	15	0	1	58	0
Rhizosolenia setigeria	0	0	2	0	0	0	1	0
Skeletonema costatum	0	38	0	16	954	10	0	95
Striatella	5	0	0	0	0	0	0	0
Thalassiosira sp.	0	1	0	13	76	19	64	95
Unidentified centric diatoms (10 u diameter)	0	0	0	0	0	0	0	0
Total Diatoms	10	39	2	59	1030	30	123	209
TOTAL PHYTO >10u	21	444	502	252	1475	173	295	570
Small monads & flagellates (4 to 10µ diameter)	23	34	56	277	502	261	66	874
Phaeocystis clumps present				+				
PLANKTONIC PROTOZOA (ciliates,oligotrichs,tintinnids)	1	11	0	22	21	6	4	76

	MEAN PHYTOPLANKTON NUMBERS PER CM3					
GRAVING SURFACE	10-May	7-Jun	18-Jul	6-Aug	6-Sep	12-Sep
DATE	1990					
ULOTHRIX	0	0	95	38	10	
CRYPTOPHYCEAE						
Cryptomonas sp.	113	1729	285	190	145	0
EULGENOPHYCEAE						
Grouped Euglenoids	0	0	32	13	29	15
DINOFLAGELLATES (Dinophyceae)						
Amphidinium sp.	0	0	0	0	0	0
Gymnodinium sp	0	6	0	13	5	0
Gyrodinium spirale	0	19	0	0	0	0
Heterocapsa	15	0	0	0	0	0
Katodinium sp.	0	0	0	0	0	0
Micracanthodinium claytonii	1	6	0	0	0	0
Oxyrrhis marina	0	0	0	0	0	0
Prorocentrum micans	0	0	0	192	40	1
Prorocentrum minimum	4	32	63	67	1	0
Protoperidinium bipes	1	0	0	0	0	1
Scripsiella	0	0	0	0	0	0
Total Dinoflagellates	21	63	63	272	45	2
DIATOMS (Bacillariophyceae)						
Ceratulina pelagica	3	0	0	0	0	0
Chaetoceros sp.	0	0	0	35	8	0
Leptocylindrus danicus	0	0	95	0	10	0
Lithodesmium	0	0	285	95	2	3
Melosira	0	0	95	0	0	0
Naviculoid Diatoms	0	0	48	0	0	0
Rhizosolenia setigera	0	0	0	0	0	0
Skeletonema sp.	8	0	0	0	49	19
Striatella	0	0	0	0	0	0
Thalassiosira sp.	32	0	0	156	14	5
Unidentified centric diatoms (10 u diameter)	0	0	0	0	0	0
Total Diatoms	43	0	523	286	83	27
TOTAL PHYTO >10u	177	1792	998	848	312	44
Small Flagellates (4 to 10µ diameter) Phaeocystis clumps present	134	393	617	152	78	22
PLANKTONIC PROTOZOA (ciliates,oligotrichs,tintinnids)	10	38	32	6	0	3

Appendix Table VI cont.

GRAVING DOCK 5m DATE	9-Jun 1988	21-Jun	28-Jun	12-Jul	9-Aug	23-Aug	20-Sep	17-Oct
CRYPTOPHYCEAE								
Cryptomonas sp.			0	0	1		182	191
EULGENOPHYCEAE								
Grouped Euglenoids			214	79	63		3	17
DINOFLAGELLATES (Dinophyceae)								
Amphidinium sp.			0	3	0		4	10
Donophysis			0	0	0		2	0
Gymnodinium sp			364	1048	0		18	17
Gyrodinium spirale			0	0	0		37	7
Micracanthodinium claytonii			0	0	0		0	18
Prorocentrum minimum			0	0	0		0	2
Protoperidinium bipes			0	0	0		0	0
Scripsiella			0	0	0		0	1
Total Dinoflagellates			364	1051	0		61	55
DIATOMS (Bacillariophyceae)								
Coscindiscus sp.			0	0	0		10	4
Chaetocerous sp.			0	0	0		6	13
Leptocylindricus sp.			0	0	0		0	4
Melosira			0	0	8		0	0
Naviculoid Diatoms			0	0	1		1	0
Rhizosolenia setigera			0	0	0		0	5
Striatella			0	0	1		0	0
Thalassiosira sp.			0	0	0		10	145
Unidentified centric diatoms (10 u diameter)			0	35	0		0	0
Total Diatoms			0	35	10		17	171
TOTAL PHYTO >10u			578	1165	74		263	434
Small Flagellates (4 - 10 µm diameter) Phaeocystis clumps present			122	177	58		28	151
PLANKTONIC PROTOZOA (ciliates,oligotrichs,tintinnids)			5	7	0		8	13

GRAVING 5m	PHYTOPLANKTON NUMBERS PER CM3		
DATE	14-Nov	19-Dec	19-Jan 1989
CRYPTOPHYCEAE			
Cryptomonas sp.	14	5	45
EULGENOPHYCEAE			
Grouped Euglenoids	1	13	10
DINOFLAGELLATES (Dinophyceae)			
Amphidinium sp.	0	11	0
Gonyaulax tamarensis	0	9	0
Gymnodinium sp	1	0	1
Gyrodinium spirale	0	0	0
Heterocapsa triquetra	0	0	0
Katodinium sp.	0	0	0
Micracanthodinium claytonii	3	0	0
Prorocentrum minimum	0	0	0
Protoperidinium bipes	1	0	0
Scripsiella	0	0	0
Total Dinoflagellates	5	20	1
DIATOMS (Bacillariophyceae)			
Chaetoceros sp.	0	0	1
Leptocylindrus danicus	1	0	0
Lithodesmium	0	0	0
Melosira	0	0	0
Naviculoid Diatoms	0	3	0
Rhizosolenia setigera	0	0	0
Skeletonema sp.	0	0	0
Thalassiosira sp.	3	1	0
Unidentified centric diatoms (10 u diameter)	0	0	0
Total Diatoms	4	4	1
TOTAL PHYTO >10u	24	42	57
Small Flagellates (4 to 10u diameter) Phaeocystis clumps present	25	29	30
PLANKTONIC PROTOZOA (ciliates,oligotrichs,tintinnids)	6	13	8

Appendix Table VI cont.

GRAVING 9m	MEAN PHYTOPLANKTON NUMBERS PER CM3					
DATE	21-Jun	28-Jun	12-Jul	9-Aug	20-Sep	17-Oct
CRYPTOPHYCEAE						
Cryptomonas sp.	0	0	0	0	148	41
EULGENOPHYCEAE						
Grouped Euglenoids	0	29	11	1	0	15
DINOFLAGELLATES (Dinophyceae)						
Amphidinium sp.	0	0	0	0	12	15
Gonyaulax tamarensis	0	0	0	0	0	0
Gymnodinium sp	12	85	241	4	15	29
Gyrodinium spirale	0	0	1	0	23	7
Heterocapsa triquetra	0	0	0	0	0	0
Micracanthodinium claytonii	0	0	0	0	0	18
Oxyrrhis marina	0	0	0	72	1	0
Prorocentrum minimum	0	6	0	0	0	8
Protoperidinium bipes	0	0	0	0	0	0
Scripsiella	0	0	0	0	0	0
Total Dinoflagellates	12	91	242	76	51	77
DIATOMS (Bacillariophyceae)						
Ceratulina	0	0	0	0	0	3
Coscinodiscus sp.	0	0	0	0	16	0
Chaetoceros sp.	0	0	0	0	4	14
Leptocylindrus danicus	0	0	0	0	0	0
Naviculoid Diatoms	18	0	0	0	0	0
Rhizosolenia setigera	0	0	0	0	0	3
Skeletonema costatum	0	0	0	0	0	0
Striatella	0	0	0	0	0	1
Thalassiosira sp.	0	0	1	0	18	185
Unidentified centric diatoms (10 u diameter)	0	0	10	8	0	0
Total Diatoms	18	0	11	8	22	203
TOTAL PHYTO >10u	30	120	264	85	221	336
Small monads & flagellates (4 to 10u diameter)	3575	42	20	37	23	129
Phaeocystis clumps present						
PLANKTONIC PROTOZOA (ciliates,oligotrichs,tintinnids)	0	0	15	62	0	9

Appendix Table VII

ALBERT SURFACE	MEAN PHYTOPLANKTON NUMBERS PER CM3							
DATE	9-Jun 1988	21-Jun	12-Jul	9-Aug	23-Aug	20-Sep	17-Oct	14-Nov
CRYPTOPHYCEAE								
Cryptomonas sp.	0	0	0	0	0	0	1468	2
EULGENOPHYCEAE								
Grouped Euglenoids	3856	1434	500	37	25	10	10	4
DINOFLAGELLATES (Dinophyceae)								
Amphidinium sp.	0	0	0	0	0	0	4	0
Gymnodinium sp	1068	5513	51	0	6	4	17	1
Gyrodinium spirale	0	0	0	0	0	149	27	0
Katodinium sp.	0	0	0	0	0	0	0	0
Micracanthodinium claytonii	0	0	0	184	69	0	178	1
Prorocentrum micans	0	0	0	2	15	0	6	1
Prorocentrum minimum	18	0	137	210	2560	0	27	3
Protoperidinium bipes	0	0	0	0	0	0	6	5
Scripsiella sp.	0	0	0	13	5	0	38	0
Total Dinoflagellates	1086	5513	188	409	2655	153	299	9
DIATOMS (Bacillariophyceae)								
Chaetoceros sp.	0	0	0	8	87	4	0	0
Thalassiosira sp.	0	0	0	0	0	0	32	8
Rhizosolenia setigera	0	0	0	35	0	0	1	0
Leptocylindrus danicus	0	0	0	0	0	0	10	0
Naviculoid Diatoms	0	25	7	84	0	0	3	1
Unidentified centric diatoms (10 u diameter)	436	8	15	3523	92	265	14	7
Total Diatoms	436	33	22	3642	179	269	60	15
TOTAL PHYTO >10µm	5378	6980	710	4298	2859	432	1921	30
Small monads & flagellates (4 to 10µm diameter)	1181	2200	6415	4642	768	15	2219	29
PLANKTONIC PROTOZOA (ciliates,oligotrichs,tintinnids)	0	433	0	38	43	178	41	0

Appendix Table VII cont.

ALBERT SURFACE	MEAN PHYTOPLANKTON NUMBERS PER CM3							
DATE	19-Dec	19-Jan 1989	22-Feb	15-Mar	17-Apr	17-May	15-Jun	28-Jun
CRYPTOPHYCEAE								
Cryptomonas sp.	45	35	158	13452	143	1	35	298
EULGENOPHYCEAE								
Grouped Euglenoids	1	2	5	0	4	3	0	1
DINOFLAGELLATES (Dinophyceae)								
Amphidinium sp.	1	0	3	0	0	0	0	0
Gonyaulax tamarensis	0	0	0	0	0	3	0	0
Gymnodinium sp	0	0	0	32	0	2	0	0
Gyrodinium spirale	0	0	0	0	0	0	0	3
Katodinium sp.	0	0	1	0	8	0	0	4
Micracanthodinium claytonii	0	0	0	15	1	0	1	1
Prorocentrum minimum	1	0	0	5	10	71	1	8
Protoperidinium bipes	1	0	2	5	1	0	0	0
Scripsiella sp.	0	0	0	56	4	0	0	0
Total Dinoflagellates	3	0	6	113	24	76	2	16
DIATOMS (Bacillariophyceae)								
Chaetoceros sp.	0	0	9	585	0	0	27	107
Thalassiosira sp.	2	4	19	1240	0	1	3	76
Rhizosolenia setigera	0	0	0	5	0	0	1	10
Leptocylindrus danicus	0	0	1	2295	0	0	578	11
Skeletonema costatum	0	0	10	1138	6	0	0	0
Naviculoid Diatoms	1	0	0	22	0	3	0	0
Unidentified centric diatoms (10 µm diameter)	0	0	15	0	1	1	0	0
Total Diatoms	2	4	54	5285	7	5	609	204
Coccolithophorids	0	0	0	0	0	5	0	0
TOTAL PHYTO >10u	48	41	223	6740	178	90	646	519
Small monads & flagellates (4 to 10u diameter)	14	10	52	41552	132	57	284	130
Phaeocystis clumps present ?			+	+				
PLANKTONIC PROTOZOA (ciliates,oligotrichs,tintinnids)	0	0	11	34	78	32	14	6

ALBERT SURFACE	**MEAN PHYTOPLANKTON NUMBERS PER CM3**							
DATE	12-Jul 1989	26-Jul	9-Aug	23-Aug	19-Sep	11-Oct	15-Nov	13-Dec
CRYPTOPHYCEAE								
Cryptomonas sp.	82	1072	76	94	44	18	49	9
EULGENOPHYCEAE								
Grouped Euglenoids	0	0	0	0	3	0	5	0
DINOFLAGELLATES (Dinophyceae)								
Amphidinium sp.	0	0	0	0	0	0	1	0
Gymnodinium sp	0	25	13	0	0	0	0	0
Gyrodinium spirale	6	3	6	0	0	0	0	0
Katodinium sp.	0	0	0	3	0	0	0	0
Prorocentrum minimum	747	99	82	0	0	0	0	4
Protoperidinium bipes	0	0	0	0	0	0	0	0
Total Dinoflagellates	753	127	101	3	0	0	1	4
DIATOMS (Bacillariophyceae)								
Chaetoceros sp.	13	0	13	10	0	0	0	0
Thalassiosira sp.	13	8	431	0	0	0	0	0
Rhizosolenia setigera	0	0	26	23	0	0	0	0
Leptocylindrus danicus	0	33	804	1176	66	0	0	0
Lithodesmium	0	0	0	213	0	0	0	0
Skeletonema costatum	0	271	0	512	2065	35	0	0
Naviculoid Diatoms	70	8	6	10	0	0	3	1
Total Diatoms	96	320	1280	1944	2131	35	3	1
TOTAL PHYTO >10µm	931	1519	1457	2041	2178	53	58	14
Small Flagellates (4 to 10µm diameter)	969	1918	551	180	66	34	29	20
Phaeocystis clumps present ?		+						
PLANKTONIC PROTOZOA (ciliates,oligotrichs,tintinnids)	19	20	38	8	17	9	1	0

ALBERT SURFACE	**MEAN PHYTOPLANKTON NUMBERS PER CM3**							
DATE	23-Jan 1990	14-Feb	13-Mar	29-Mar	6-Apr	24-Apr	3-May	10-May
CRYPTOPHYCEAE								
Cryptomonas sp.	41	39	460	149	117	83	285	7
EULGENOPHYCEAE								
Grouped Euglenoids	0	0	2	81	0	98	76	0

Appendix Table VII cont.

DINOFLAGELLATES (Dinophyceae)								
Gymnodinium sp	0	1	0	0	0	6	57	0
Heterocapsa triquetra	0	0	2	0	0	6	19	20
Katodinium sp.	0	0	0	0	0	0	19	0
Oxyrrhis marina	0	1	0	0	0	0	0	0
Prorocentrum minimum	0	1	0	0	0	0	0	10
Protoperidinium bipes	0	0	0	0	0	0	19	0
Total Dinoflagellates	0	3	2	0	0	12	114	30
DIATOMS (Bacillariophyceae)								
Coscinodiscus sp.	0	0	0	0	10	13	0	0
Thalassiosira sp.	0	0	0	106	40	0	0	7
Leptocylindrus danicus	0	0	0	0	0	0	0	14
Skeletonema costatum	0	0	72	608	2166	0	0	0
Naviculoid Diatoms	0	0	19	5	0	0	0	0
Total Diatoms	0	0	91	719	2216	13	0	21
TOTAL PHYTO >10µm	41	42	555	949	2333	206	475	58
Small monads & flagellates (4 to 10µm diameter)	23	56	851	717	859	906	874	1154
PLANKTONIC PROTOZOA (ciliates,oligotrichs,tintinnids)	0	3	4	5	8	0	76	5

ALBERT SURFACE	PHYTOPLANKTON NUMBERS PER CM3			
DATE	7-Jun 1990	18-Jul	9-Aug	12-Sep
Ulothrix subflacca	0	19	0	0
CRYPTOPHYCEAE				
Cryptomonas sp.	154	19	14	0
EULGENOPHYCEAE				
Grouped Euglenoids	0	0	4	1
DINOFLAGELLATES (Dinophyceae)				
Gymnodinium sp	0	19	0	0
Prorocentrum micans	0	0	0	6
Prorocentrum minimum	0	127	0	0
Total Dinoflagellates	0	146	0	6
DIATOMS (Bacillariophyceae)				
Chaetoceros sp.	3	127	80	0
Thalassiosira sp.	18	44	15	8
Leptocylindrus danicus	0	76	0	0
Lithodesmium undulatum	0	412	1	0
Skeletonema costatum	171	0	10	0
Naviculoid Diatoms	13	0	0	0
Total Diatoms	205	659	106	8
TOTAL PHYTO >10u	359	843	124	15
Small monads & flagellates (4 to 10u diameter)	812	1685	182	6
PLANKTONIC PROTOZOA (ciliates,oligotrichs,tintinnids)	30	19	4	0

Appendix Table VII cont.

ALBERT 5m	MEAN PHYTOPLANKTON NUMBERS PER CM3				
DATE	12-Jul	9-Aug	20-Sep	17-Oct	14-Nov
	1988				
CRYPTOPHYCEAE					
Cryptomonas sp.	8	0	7	44	0
EULGENOPHYCEAE					
Grouped Euglenoids	32	0	0	3	1
DINOFLAGELLATES (Dinophyceae)					
Amphidinium sp.	0	0	0	7	0
Gymnodinium sp	25	38	19	24	0
Gyrodinium spirale	8	8	153	18	0
Micracanthodinium claytonii	0	68	0	8	0
Prorocentrum minimum	0	0	3	1	1
Scripsiella sp.	0	0	6	0	6
Total Dinoflagellates	33	114	181	58	7
DIATOMS (Bacillariophyceae)					
Chaetoceros sp.	0	0	0	24	0
Coscindiscus sp	0	0	8	18	0
Thalassiosira sp.	0	0	24	123	4
Rhizosolenia setigera	0	2	0	1	0
Leptocylindrus danicus	0	0	0	18	0
Striatella sp.	0	0	0	3	0
Skeletonema sp.	0	13	0	0	0
Naviculoid Diatoms	40	0	0	0	0
Unidentified centric diatoms (10 u diameter)	57	84	0	0	0
Total Diatoms	97	99	32	186	4
TOTAL PHYTO >10µm	170	213	220	291	12
Small Monads & flagellates (4 to 10µm diameter)	184	663	67	52	27
PLANKTONIC PROTOZOA (ciliates,oligotrichs,tintinnids)	0	58	195	8	1

ALBERT 5m	MEAN PHYTOPLANKTON NUMBERS PER CM3						
DATE	19-Dec 1989	19-Jan	22-Feb	15-Mar	17-Apr	17-May	15-Jun 28-Jun
CRYPTOPHYCEAE							
Cryptomonas sp.	57	60	69	1051	80	0	367
EULGENOPHYCEAE							
Grouped Euglenoids	0	1	3	51	44	0	0
DINOFLAGELLATES (Dinophyceae)							
Amphidinium sp.	1	3	6	0	40	10	7
Gonyaulax tamarensis	0	0	0	0	0	0	0
Gymnodinium sp	0	4	4	51	35	0	13
Gyrodinium spirale	0	0	1	0	0	11	3
Heterocapsa triquetra	0	0	0	63	0	0	0
Katodinium sp.	0	0	0	0	0	0	0
Micracanthodinium claytonii	0	1	0	0	6	0	10
Prorocentrum minimum	1	0	0	38	0	5	29
Protoperidinium bipes	0	0	1	0	0	0	0
Scripsiella sp.	1	0	0	25	0	0	0
Total Dinoflagellates	3	8	12	177	81	26	62
DIATOMS (Bacillariophyceae)							
Chaetoceros sp.	0	0	0	63	0	0	35
Thalassiosira sp.	1	0	30	760	19	0	109
Rhizosolenia setigera	0	0	0	0	0	0	0
Leptocylindrus danicus	0	0	4	1609	24	0	16
Skeletonema costatum	0	0	34	0	0	0	13
Naviculoid Diatoms	0	0	0	0	0	0	6
Total Diatoms	1	0	68	2432	43	0	179
TOTAL PHYTO >10u	61	69	152	3711	248	26	608
Small monads & flagellates (4 to 10u diameter)	20	34	123	41570	219	39	263
Phaeocystis clumps present ?				+			
PLANKTONIC PROTOZOA (ciliates,oligotrichs,tintinnids)	6	8	11	0	17	38	22
ROTIFERS						1	

ALBERT 5m	PHYTOPLANKTON NUMBERS PER CM3							
DATE	12-Jul 1989	26-Jul	9-Aug	23-Aug	19-Sep	11-Oct	15-Nov	13-Dec
CRYPTOPHYCEAE								
Cryptomonas sp.	29	104		46	23			
DINOFLAGELLATES (Dinophyceae)								
Amphidinium sp.	0	10		0	0			
Gymnodinium sp	3	29		5	3			
Gyrodinium spirale	10	23		0	0			
Micracanthodinium claytonii	0	6		0	0			
Oxyrrhis marina	0	41		0	0			
Prorocentrum minimum	140	1		0	0			
Protoperidinium bipes	0	0		9	0			
Total Dinoflagellates	153	110		14	3			
DIATOMS (Bacillariophyceae)								
Chaetoceros sp.	13	68		10	4			
Thalassiosira sp.	22	11		70	11			
Rhizosolenia setigera	3	0		8	0			
Leptocylindrus danicus	0	13		839	6			
Lithodesmium	0	1		359	0			
Skeletonema costatum	57	38		31	1496			
Naviculoid Diatoms	137	5		0	0			
Total Diatoms	232	136		1317	1517			
TOTAL PHYTO >10µm	414	350		1377	1543			
Small monads &fFlagellates (4 to 10µm diameter)	120	110		223	8			
PLANKTONIC PROTOZOA (ciliates,oligotrichs,tintinnids)	7	27		0	0			

Appendix Table VIII

QUEENS SURFACE	MEAN PHYTOPLANKTON NUMBERS PERCM3							
DATE	21-Jun 1988	12-Jul	9-Aug	23-Aug	20-Sep	17-Oct	14-Nov	19-Jan 1989
CRYPTOPHYCEAE								
Cryptomonas sp.	0	162	0	0	519	602	9	71
EULGENOPHYCEAE								
Grouped Euglenoids	3565	1397	65	120	13	32	6	20
DINOFLAGELLATES (Dinophyceae)								
Amphidinium sp.	0	0	0	4	0	0	0	0
Gonyaulax tamarensis	0	0	0	0	0	0	0	0
Gymnodinium sp	11515	95	255	4	13	5	10	11
Gyrodinium spirale	0	0	0	0	120	0	0	0
Heterocapsa triquetra	0	10	65	18	0	5	6	0
Katodinium sp.	0	0	0	0	0	5	0	0
Micracanthodinium claytonii	0	29	60	123	13	85	3	0
Oxyrrhis marina	0	0	0	0	0	0	0	0
Prorocentrum micans	0	0	0	25	0	0	1	0
Prorocentrum minimum	0	67	41	490	0	0	5	4
Protoperidinium bipes	0	0	0	0	0	0	0	0
Scripsiella sp.	0	0	0	0	0	0	22	0
Total Dinoflagellates	11515	201	421	639	146	100	47	15
DIATOMS (Bacillariophyceae)								
Chaetoceros sp.	0	0	0	7	0	111	0	0
Thalassiosira sp.	0	0	0	0	63	296	11	1
Rhizosolenia setigera	0	19	0	0	0	11	0	0
Leptocylindrus danicus	0	0	0	0	0	0	0	0
Lithodesmium undulatum	0	0	10	0	0	0	0	0
Melosira sp.	0	0	0	0	0	37	0	0
Striatella sp.	0	0	0	0	0	0	0	0
Skeletonema costatum	0	0	0	0	0	0	0	0
Naviculoid Diatoms	23	95	0	5	0	0	5	0
Unidentified centric diatoms (10 µm diameter)	0	247	6378	99	0	0	27	1
Total Diatoms	23	361	6388	111	63	455	43	2
TOTAL PHYTO >10µm	15103	2121	6874	870	741	1189	107	109
Small monads & flagellates (4 to 10µm diameter)	99400	1986	947	1372	279	1214	144	127
Phaeocystis clumps present								
PLANKTONIC PROTOZOA (ciliates,oligotrichs,tintinnids)	402	67	125	47	13	52	9	14

Appendix Table VIII cont.

MEAN PHYTOPLANKTON NUMBERS PERCM3

QUEENS SURFACE								
DATE	22-Feb 1989	15-Mar	17-Apr	17-May	15-Jun	28-Jun	12-Jul	26-Jul
CRYPTOPHYCEAE								
Cryptomonas sp.	134	722	746	7	260	190	0	25
EULGENOPHYCEAE								
Grouped Euglenoids	7	89	0	0	1	0	0	0
DINOFLAGELLATES (Dinophyceae)								
Amphidinium sp.	0	0	26	0	0	0	0	0
Dinophysis recurva	0	0	0	0	0	0	25	13
Gonyaulax tamarensis	2	0	53	0	0	0	0	0
Gymnodinium sp	18	76	131	0	17	81	38	215
Gyrodinium spirale	0	0	0	0	0	0	0	0
Heterocapsa triquetra	22	38	108	0	4	36	0	0
Katodinium sp.	0	0	274	0	6	0	0	0
Micracanthodinium claytonii	2	38	151	0	0	0	38	0
Oxyrrhis marina	0	0	0	0	0	0	0	0
Prorocentrum micans	0	0	0	0	0	0	0	0
Prorocentrum minimum	0	63	274	1001	30	5646	20385	26403
Protoperidinium bipes	35	0	0	3	5	0	0	0
Scripsiella sp.	0	0	0	0	0	0	0	0
Total Dinoflagellates	79	215	1017	1004	62	5763	20486	26631
DIATOMS (Bacillariophyceae)								
Chaetoceros sp.	0	139	0	0	0	0	0	0
Thalassiosira sp.	31	824	283	7	0	76	13	0
Rhizosolenia setigera	0	0	0	0	0	25	0	0
Leptocylindrus danicus	90	2825	0	0	0	36	0	0
Lithodesmium undulatum	0	0	0	0	0	0	0	0
Melosira sp.	8	0	0	0	0	0	0	0
Striatella sp.	0	0	0	0	0	0	0	0
Skeletonema costatum	12	279	0	0	0	0	0	0
Naviculoid Diatoms	3	13	0	0	1	25	13	0
Unidentified centric diatoms (10 µm diameter)	0	0	17	0	3	0	0	13
Total Diatoms	144	4080	300	7	4	162	26	13
TOTAL PHYTO >10µm	364	5109	2143	1018	327	6115	20512	26669
Small monads & flagellates (4 to 10µm diameter)	1769	53181	61636	54	466	2697	177	228
Phaeocystis clumps present						+		
PLANKTONIC PROTOZOA (ciliates,oligotrichs,tintinnids)	2	25	280	3	79	503	165	380

QUEENS SURFACE	MEAN PHYTOPLANKTON NUMBERS PERCM3							
DATE	9-Aug 1989	23-Aug	19-Sep	11-Oct	15-Nov	13-Dec	23-Jan 1990	14-Feb
Ulothrix subflaccida	649	3079	549	10	0	0	0	0
CRYPTOPHYCEAE								
Cryptomonas sp.	533	513	655	75	98	29	48	70
EULGENOPHYCEAE								
Grouped Euglenoids	0	6	21	3	4	3	0	0
DINOFLAGELLATES (Dinophyceae)								
Amphidinium sp.	0	0	0	0	0	0	0	0
Gonyaulax tamarensis	0	0	0	0	0	0	0	0
Gymnodinium sp	63	22	0	0	1	4	0	0
Gyrodinium spirale	0	0	0	3	0	0	0	0
Heterocapsa triquetra	0	0	0	3	0	1	0	0
Katodinium sp.	0	0	0	0	0	0	0	0
Micracanthodinium claytonii	37	0	0	0	0	0	0	0
Oxyrrhis marina	0	0	0	0	0	0	3	4
Prorocentrum micans	0	6	0	0	1	0	0	0
Prorocentrum minimum	4661	32	42	2	0	1	0	0
Protoperidinium bipes	0	0	0	0	0	0	0	0
Scripsiella sp.	21	0	0	0	0	0	0	0
Total Dinoflagellates	4782	60	42	8	2	6	3	4
DIATOMS (Bacillariophyceae)								
Chaetoceros sp.	449	215	2344	8	3	0	0	0
Thalassiosira sp.	486	6	0	0	0	0	0	0
Rhizosolenia setigera	37	26	0	0	0	1	0	0
Leptocylindrus danicus	1293	0	0	0	0	0	0	0
Lithodesmium undulatum	21	513	0	0	1	0	0	1
Melosira sp.	0	0	0	0	0	0	0	0
Striatella sp.	0	0	0	0	0	0	0	0
Skeletonema costatum	4614	57	20208	1938	0	0	25	0
Naviculoid Diatoms	470	6	0	0	0	0	0	20
Unidentified centric diatoms (10 µm diameter)	0	57	0	0	0	0	0	0
Total Diatoms	7370	880	22552	1946	4	1	25	21
TOTAL PHYTO >10µm	13334	4538	23819	2042	108	39	76	95
Small monads & flagellates (4 to 10µm diameter) Phaeocystis clumps present	34795	317	591	104	74	107	25	10
PLANKTONIC PROTOZOA (ciliates,oligotrichs,tintinnids)	264	108	170	5	10	4	4	9

Appendix Table VIII cont.

QUEENS SURFACE	MEAN PHYTOPLANKTON NUMBERS PERCM3						
DATE	13-Mar 1990	29-Mar	6-Apr	20-Apr	24-Apr	28-Apr	3-May	10-May
Eulothrix subflaccida	0	0	0	0	1229	0	0
CRYPTOPHYCEAE							
Cryptomonas sp.	785	215	716	963	3003	456	76
EULGENOPHYCEAE							
Grouped Euglenoids	87	13	13	0	0	57	0
DINOFLAGELLATES (Dinophyceae)							
Amphidinium sp.	0	0	0	0	0	0	0
Gonyaulax tamarensis	0	0	0	0	0	0	0
Gymnodinium sp	38	13	95	1178	342	0	25
Gyrodinium spirale	0	0	0	0	0	0	0
Heterocapsa triquetra	0	0	6	25	165	1254	146
Katodinium sp.	0	0	0	0	13	95	51
Micracanthodinium claytonii	0	0	0	0	0	0	6
Oxyrrhis marina	0	0	0	0	0	0	0
Prorocentrum micans	0	0	0	0	0	0	0
Prorocentrum minimum	0	0	0	38	13	0	38
Protoperidinium bipes	13	0	0	0	25	19	0
Scripsiella sp.	13	0	0	0	0	0	6
Total Dinoflagellates	64	13	101	1241	558	1368	272
DIATOMS (Bacillariophyceae)							
Ceratulina	0	0	0	0	0	38	583
Coscinodiscus sp.	0	0	0	0	13	0	0
Chaetoceros sp.	684	0	0	279	342	0	0
Thalassiosira sp.	166	13	133	139	355	0	608
Rhizosolenia setigera	0	0	0	0	63	0	0
Leptocylindrus danicus	0	0	0	13	63	0	0
Lithodesmium undulatum	0	0	0	0	0	0	0
Melosira sp.	0	0	0	0	0	0	0
Striatella sp.	0	0	0	0	0	0	0
Skeletonema costatum	4713	95	139	177	1406	0	89
Naviculoid Diatoms	0	0	0	0	0	0	0
Unidentified centric diatoms (10 µm diameter)	0	0	32	0	0	0	0
Total Diatoms	5563	108	304	608	2242	38	1280
TOTAL PHYTO >10µm	6499	349	1134	2812	7032	1919	1628
Small monads & flagellates (4 to 10µm diameter) Phaeocystis clumps present	6448	9354	6040	5966	4789	2395	2813
PLANKTONIC PROTOZOA (ciliates,oligotrichs,tintinnids)	13	25	13	51	25	475	6

QUEENS SURFACE	MEAN PHYTOPLANKTON NUMBERS PERCM3			
DATE	7-Jun 1990	18-Jul	9-Aug	12-Sep
CRYPTOPHYCEAE				
Cryptomonas sp.	823	0	18	54
EULGENOPHYCEAE				
Grouped Euglenoids	6	0	0	28
DINOFLAGELLATES (Dinophyceae)				
Amphidinium sp.	0	0	0	0
Gonyaulax tamarensis	0	0	0	0
Gymnodinium sp	0	124	0	0
Gyrodinium spirale	6	0	0	0
Heterocapsa triquetra	6	41	0	0
Katodinium sp.	0	0	6	0
Micracanthodinium claytonii	6	130	3	0
Oxyrrhis marina	0	0	0	0
Prorocentrum micans	0	41	1304	103
Prorocentrum minimum	44	28094	479	42
Protoperidinium bipes	0	16	6	0
Scripsiella sp.	0	51	3	0
Total Dinoflagellates	56	28497	1801	145
DIATOMS (Bacillariophyceae)				
Chaetoceros sp.	13	25	0	77
Thalassiosira sp.	0	0	3	85
Rhizosolenia setigera	0	0	0	0
Leptocylindrus danicus	0	0	0	0
Lithodesmium undulatum	6	209	0	10
Melosira sp.	0	0	0	0
Striatella sp.	0	0	0	0
Skeletonema costatum	310	0	71	3497
Naviculoid Diatoms	0	0	0	10
Unidentified centric diatoms (10 µm diameter)	0	0	0	0
Total Diatoms	329	234	74	3679
TOTAL PHYTO >10µm	1214	28731	1893	3906
Small monads & flagellates (4 to 10µm diameter)	30600	747	306	1155
Phaeocystis clumps present +				
PLANKTONIC PROTOZOA (ciliates,oligotrichs,tintinnids)	32	25	12	10

Appendix Table IX

GRAVING DOCK	ZOOPLANKTON - NUMBERS / m2							
DATE 1988	21-Jun	28-Jun	12-Jul	27-Jul	9-Aug	23-Aug	20-Sep	14-Nov
Annelida	140µ	140µ	¡40µ	140µ				
Polydora ciliata	245810	147101	1136	400	112	4	0	
Unid. errant polycheate	0	0	0	0	0	0	0	1
Crustacea								
Barnacle nauplii	0	0	32	0	37	28	0	0
Podon sp.	0	0	44	0	0	0	0	0
Copepoda :								
Copepod nauplii	0	85	2494	0	3	0	0	0
Copepodites	0	0	1763	0	0	0	0	0
Eurytemora affinis	476	19	1227	44700	33878	7750	15	<1
Acartia clausii	0	0	0	865	234	0	<1	0
Tisbe longicornis	0	0	23	0	31	0	0	2
Ctenophora								
Pleurobrachia pileus	0	0	4	0	13	9	2	0
Total Zooplankton	246286	147205	6723	45965	34295	7791	19	4

21st June to 27th July 140µm net mesh used, 250 µm mesh used at all other times

1989 DATE	19-Jan	22-Feb	15-Mar	17-Apr	17-May	21-Jun	28-Jun	12-Jul
Annelida								
Polydora ciliata	0	0	0	<1	0	0	<1	<1
Cnidaria								
Aurelia aurita	0	1	18	13	15	1	0	0
Crustacea								
Barnacle nauplii	0	1	0	5	0	0	0	0
Podon sp.	0	0	0	0	<1	<1	0	2
Copepoda								
Eurytemora affinis	0	<1	0	0	0	0	0	0
Tisbe longicornis	<1	0	1	5	3	2	0	0
Tigriopus brevicornis	<1	0	0	6	6	5	1	0
Total Zooplankton	<2	3	19	29	24	8	1	2

DATE 1989	26-Jul	9-Aug	23-Aug	19-Sep	15-Nov	23-Jan 1990	14-Feb	13-Mar
Annelida								
Polydora ciliata	<1	0	0	0	0	0	0	0
Cnidaria								
Anthozoan larvae	0	<1	22	<1	0	0	0	0
Small Aurelia medusae	0	0	0	0	0	1	3	2
Crustacea								
Barnacle nauplii	0	0	0	0	0	0	0	1
Podon sp.	2	5	0	0	0	0	0	0
Copepoda								
Tisbe longicornis	0	1	0	0	<1	<1	2	0
Tigriopus brevicornis	1	4	2	<1	1	0	0	0
Dinophyceae								
Noctiluca scintillans	0	0	0	1	0	0	0	0
Total Zooplankton	3	10	24	<2	<2	<2	5	3

GRAVING DOCK	ZOOPLANKTON - NUMBERS / m^2					
DATE	19-Apr	10-May	7-Jun	3-Jul	9-Aug	12-Sep
Annelida	1990					
Polydora ciliata	0	2	0	0	0	0
Cnidaria						
Obelia sp.	0	2	0	0	0	0
Aurelia aurita	35	6	0	0	0	0
Sarsia sp.	2	2	0	0	<1	0
Crustacea						
Barnacle nauplii	1	5	0	0	0	<1
Carcinus zoea	0	0	0	0	<1	0
Copepoda						
Eurytemora affinis	0	1	<1	0	<1	0
Tisbe longicornis	3	7	<1	2	0	0
Tigriopus brevicornis	3	19	5	8	<1	<1
Ascidiacea larvae	0	0	0	0	2	0
Total Zooplankton	44	44	6	10	4	<2

Appendix Table X

ALBERT DOCK	ZOOPLANKTON - NUMBERS / m2							
DATE 1988	21-Jun	28-Jun	12-Jul	27-Jul	9-Aug	23-Aug	20-Sep	14-Nov
Annelida	140µ	140µ	140µ	140µ				
Polydora ciliata	1288		92481		74	1	0	0
Errant polycheate	0		0		0	0	1	0
Cnidaria								
Aurelia aurita	0		0		4	0	0	0
Crustacea								
Barnacle naupl./cypr.	54		58		1368	6	1	0
Copepoda								
Copepod nauplii	81		72		0	0	0	0
Eurytemora affinis	0		14580		7805	56	10	0
Acartia clausii	0		0		61	0	0	0
Tisbe longicornis	0		0		9	0	0	4
Podon sp.	54		8954		0	0	0	0
Ctenophora								
Pleurobrachia pileus	0		0		0	0	1	0
Mollusca								
Mytilus	0		0		9	0	0	0
Total zooplankton	1477		116145		9330	57	13	4

21st June and 12th July140 µm mesh net used. All other dates 250 µm mesh net used.

DATE 1989	19-Jan	22-Feb	15-Mar	17-Apr	17-May	21-Jun	28-Jun	12-Jul
Annelida								
Polydora ciliata	0	0	0	0	0	0	1	1
Cnidaria								
Aurelia medusae	0	1	93	29	6	0	0	0
Crustacea								
Podon sp.	1	0	0	0	4	0	0	2
Copepoda								
Copepodites	1	0	0	0	0	0	0	0
Tisbe longicornis	3	5	3	0	0	0	1	2
Tigriopus brevicornis	0	0	2	1	3	1	2	1
Total zooplankton	5	6	98	30	13	1	4	6

DATE 1989	26-Jul	9-Aug	23-Aug	19-Sep	11-Oct	15-Nov	23-Jan 1990	14-Feb
Cnidaria								
Anthozoan larvae	0	0	1	0	0	0	0	0
Crustacea								
Barnacle nauplii	0	1	4	0	0	0	0	0
Podon sp.	2	2	3	0	0	0	0	0
Copepoda								
Acartia clausii	0	0	0	2	0	0	0	0
Tigriopus brevicornis	1	0	1	0	0	1	0	0
Tisbe longicornis	0	0	0	0	1	0	0	2
Mollusca								
Mytilus edulis	0	0	0	1	0	0	0	0
Dinophyceae								
Noctiluca scintillans	0	0	0	0	2	15	0	0
Total zooplankton	3	3	9	3	1	1	0	2

Appendix Table X

ALBERT DOCK	ZOOPLANKTON - NUMBERS / m2						
DATE 1990	13-Mar	19-Apr	10-May	7-Jun	3-Jul	9-Aug	12-Sep
Annelida							
Polydora ciliata	0	1	0	1	1	0	0
Cnidaria							
Aurelia aurita	0	6	1	0	0	0	0
Obelia sp.	0	0	1	0	0	0	0
Crustacea							
Barnacle nauplii	0	4	203	3	5	0	0
Carcinus zoea	0	0	3	1	0	9	0
Podon sp.	0	0	1	0	0	0	0
Copepoda							
Acartia clausii	0	0	2	0	2	1	1
Eurytemora affinis	0	0	14	1	24	1	0
Tigriopus brevicornis	0	0	9	1	5	0	0
Tisbe longicornis	0	10	3	0	0	0	1
Ascidiacea larvae	0	0	0	0	0	1	2
Total zooplankton	0	21	237	7	37	12	4

Appendix Table XI

QUEENS DOCK	ZOOPLANKTON - NUMBERS / m2							
DATE 1988	21-Jun	28-Jun	12-Jul	27-Jul	9-Aug	23-Aug	20-Sep	14-Nov
Annelida	140µ	140µ	140µ	140µ				
Polydora ciliata	34800		643			0	0	0
Errant polychaete	0		0			0	2	0
Crustacea								
Barnacle nauplii	867		736			2	2	0
Podon sp.	267		27336			0	0	0
Copepoda :								
Copepod nauplii	267		151			0	0	0
Eurytemora affinis	600		46719			38	6	2
Acartia clausii	0		0			0	0	0
Tisbe longicornis	0		10			0	0	6
Ctenophora								
Pleurobrachia pileus	0		2			0	0	0
Total Zooplankton	36801		75597			40	10	8

21st June and 12th July 140µm net mesh used, 250 µm mesh used at all other times

DATE 1989	19-Jan	22-Feb	15-Mar	17-Apr	17-May	21-Jun	28-Jun	12-Jul
Annelida								
Polydora ciliata	0	0	5	3	6	2	2	2
Errant polycaete	0	0	0	2	0	0	0	0
Cnidaria								
Aurelia aurita	0	2	22	84	6	0	0	0
Crustacea								
Barnacle nauplii	0	0	0	10	0	0	0	0
Carcinus zooea	0	0	0	0	0	0	0	2
Podon sp.	0	0	0	0	3	0	2	2
Copepoda								
Eurytemora affinis	0	0	0	0	0	0	0	0
Tisbe longicornis	0	2	2	8	0	3	0	3
Tigriopus brevicornis	0	0	0	3	11	5	0	0
Total Zooplankton	0	4	29	110	26	10	4	9

DATE 1989	26-Jul	9-Aug	23-Aug	19-Sep	15-Nov	23-Jan 1990	14-Feb	13-Mar
Annelida								
Polydora ciliata	2	2		0	0	0	0	0
Errant polychaete	0	0		2	0	0	0	0
Crustacea								
Barnacle nauplii	3	16		2	0	4	0	1
Podon sp.	0	52		0	0	0	0	0
Copepoda								
Tisbe longicornis	2	3		0	0	1	0	2
Tigriopus brevicornis	0	2		3	0	0	0	0
Dinophyceae								
Noctiluca scintillans	0	0		149	2	0	0	0
Total Zooplankton	7	75		7	0	5	0	3

QUEENS DOCK	ZOOPLANKTON - NUMBERS / m2					
DATE 1990	19-Apr	10-May	7-Jun	3-Jul	9-Aug	12-Sep
Annelida						
Polydora ciliata	0	5	0	2	0	0
Errnt polychaete	0	0	1	5	0	0
Cnidaria						
Aurelia aurita	49	0	0	0	2	0
Sarsia sp.	0	0	0	0	0	2
Crustacea						
Barnacle nauplii	13	30	0	38	14	26
Copepoda						
Acartia affinis	2	6	0	0	0	2
Eurytemora affinis	0	6	3	0	0	0
Tisbe longicornis	19	10	5	0	0	0
Tigriopus brevicornis	3	19	14	0	3	6
Ascidiacea larvae	0	0	0	0	17	0
Dinophyceae						
Noctiluca scintillans	0	0	0	0	527	5
Total Zooplankton	86	76	23	45	36	36

App. Table XII Graving Dock - Abundance estimates of flora and fauna.

Graving Dock June 1988						
Depth in m (vertical surfaces)	0.2	1	2	3	4	7
Ephemeral green alga	F					
Bryozoa		A	A	C	F	
Molgula manhattensis						
Mytilus edulis						

Graving Dock July 1989						
Depth in m (vertical surfaces)	0.2	1	2	3	4	7
Ephemeral green alga						
Bryozoa	C	A	A	A	F	
Molgula manhattensis			P	P	P	
Mytilus edulis	F	F	F	P		

Graving Dock November1989						
Depth in m (vertical surfaces)	0.2	1	2	3	4	7
Ephemeral green alga	P	P				
Spirogyra mat						
Punctaria latifolia						
Bryozoa	F	C	F	F	F	P
Ascidiella aspersa						
Molgula manhattensis			P			
Mytilus edulis	C	C	F	F		

Graving Dock June 1990						
Depth in m (vertical surfaces)	0.2	1	2	3	4	7
Ephemeral green alga	F	F	P			
Spirogyra mat					P	
Punctaria latifolia	P	P	P			
Bryozoa	A	A	A	A	F	
Ascidiella aspersa				P		
Molgula manhattensis				P		
Mytilus edulis	C	C	F	F	P	P

Graving Dock August 1990						
Depth in m (vertical surfaces)	0.2	1	2	3	4	7
Ephemeral green alga	F					
Ceramium rubrum	P					
Spirogyra mat	C	C	C			
Bryozoa	C	C	F	P		
Botryllus schlosseri	P	P				
Molgula manhattensis	F	C	P	P	F	
Mytilus edulis	C	F	F	P		P

Key -
From % cover estimates:
Abundant (A) = >50%
Common (C) = 20 - 49 %
Frequent (F) = 5 - 19 %
Present (P) = 1 - 4 %

Notes:
Ephemeral green algae - main species are Enteromorpha intestinalis, Cladophora vagabunda and colonial diatoms.

Polydora - Mainly P. ciliata, some P.pulchra.

Nereis - species N. virens, N. pelagica, N. diversicola.

Bryozoa - mainly Conopeum seurati

Appendix Table XIII Albert Dock - Abundance estimates of wall flora and fauna.

Depth in m	Albert Dock July 1989 0.2	1	2	3	4	Albert Dock Nov. 1989 0.2	1	2	3
Ephemeral green alga	F					P			
Bryozoa		F	F	F	C				P
Molgula manhattensis			P		P				
Mytilus edulis	A	A	A	A	A	A	A	A	A

Depth in m	Albert Dock May1990 0.2	1	2	3	4	Albert Dock August 1990 0.2	1	2	3	4
Ephemeral green alga	P	P				F	P			
Ulva lactuca	F	F	F	P		C	F			
Ceramium rubrum						P	P			
Punctaria latifolia	F	F	P	P						
Bryozoa			P	P	P			P	P	
Molgula manhattensis								P		
Mytilus edulis	A	A	A	A	A	A	A	A	A	A

Depth in m	Albert Dock January 1991 0.2	1	2	3	4
Ephemeral green alga	F				
Ulva lactuca	P				
Ceramium rubrum	F		F		
Halichondria panacea		P		P	
Ciona intestinalis					P
Mytilus edulis	F	C	A	C	A

Key -
From % cover estimates:
Abundant (A) = >50%
Common (C) = 20 - 49 %
Frequent (F) = 5 - 19 %
Present (P) = 1 - 4 %

Notes:

Ephemeral green algae - main species are Enteromorph intestinalis, Cladophora vagabunda and colonial diaton

Polydora - Mainly P. ciliata, some P.pulchra.

Nereis species - N. virens, N pelagica, N. diversicola

Bryozoa - mainly Conopeum seurati

Appendix table XIV Queens Dock - Abundance estimates of wall flora and fauna with depth.

	Queens Dock June 1988				Queens Dock July 1989			
Depth in m	0.2	1	2		0.2	1	2	
Ephemeral green alga	P							
Bryozoa	A	A	A		P	A	C	
Molgula manhattensis							P	
Mytilus edulis					F	C	F	

	Queens Dock Nov.1989				Queens Dock May 1990			
Depth in m	0.2	1	2		0.2	1	2	3
Ephemeral green alga	C				F	F		
Punctaria latifolia					C			
Obelia dichotoma					F	F		
Bryozoa	C	A	P		F		P	
Botryllus schlosseri					P			
Molgula manhattensis		P						
Mytilus edulis	C	C	F		C	A	C	P
Balanus improvisus							P	P

	Queens Dock August 1990			
Depth in m	0.2	1	2	3
Ephemeral green alga	P			
Ceramium spp.	P			
Bryozoa	A	C	A	A
Botryllus schlosseri	P			
Molgula manhattensis	P	F	F	C
Mytilus edulis	C	A	C	P

Key -
From % cover estimates:
Abundant (A) = >50%
Common (C) = 20 - 49 %
Frequent (F) = 5 - 19 %
Present (P) = 0 - 4 %

Notes:
Ephemeral green algae - main species are Enteromorpha intestinalis, Cladophpora vagabunda, colonial diatoms.

Polydora - Mainly P. ciliata, some P.pulchra.

Bryozoa - mainly Conopeum seurati

Appendix Table XV Graving Dock - Fauna in 25 x 25 cm scrape.

	Graving Dock June 1988						Graving Dock August1989					
Depth in m (vertical surfaces)	0.2	1	2	3	4	7	0.2	1	2	3	4	7
Corophium insidiosum							A	C	C	C	C	
Gammarus salinus							P		P	P		
Jassa marmorata							F	P				
Microdeutopus gryllotalpa							C	F	F	P	P	
Bryozoa		√	√	√	√		√	√	√	√	√	
Balanus improvisus									√	√		
Polydora spp.		√	√	√	√		√	√	√	√	√	√
Molgula manhattensis							√	√			√	
Mytilus edulis							√	√	√	√	√	
Mytilus juv. < 15 mm		√							√	√		

	Graving Dock Nov. 1989						Graving Dock June 1990					
Depth in m (vertical surfaces)	0.2	1	2	3	4	7	0.2	1	2	3	4	7
Corophium insidiosum	F	F	F	F	F	P	C	C	C	C	C	F
Gammarus salinus	P	P	P				P			F		
Jassa marmorata								P	P		P	P
Microdeutopus gryllotalp	F	P	P	P	P		C	F	C		F	P
Bryozoa	√	√	√	√	√	√	√	√	√			√
Obelia dichotoma							√	√			√	√
Balanus improvisus	√	√	√	√	√	√					√	
Nereis spp.	√	√	√		√					√	√	
Polydora spp.				√	√		√	√	√	√	√	√
Ascidiella aspersa		√	√									
Molgula manhattensis	√	√	√	√	√	√		√		√	√	√
Metridium senile												√
Mytilus edulis	√	√	√	√	√		√	√	√	√	√	√
Mytilus juv. < 15 mm		√	√	√	√							√

	Graving Dock August 1990						Key -
Depth in m (vertical surfaces)	0.2	1	2	3	4	7	Amphipod abundance estimates:
Corophium insidiosum	C	C	C	C	C	F	Abundant (A) = >1000
Gammarus salinus							Common (C) = 100 - 999
Jassa marmorata		P					Frequent (F) = 10 - 99
Microdeutopus gryllotalp	C	F	C	C	C	P	Present = < 10
							Animals per 25 x 25 cm quadrat
Bryozoa	√	√	√		√		All other species presence
Obelia dichotoma		√					/absence only.
Balanus improvisus			√	√			
Polydora spp.	√	√	√		√	√	
Molgula manhattensis	√	√	√	√	√	√	Nereis - species present are
Metridium senile							N. virens, N. pelagica,
Mytilus edulis	√	√	√	√	√	√	N. diversicola.
Mytilus juv. < 15 mm	√		√				Polydora - Mainly P. ciliata,
							occ. P.pulchra.

App. Table XVI Albert Dock - Fauna in 25 x 25 cm wall scrapes.

	Albert Dock July 1989					Albert Dock Nov. 1989			
Depth in m	0.2	1	2	3	4	0.2	1	2	3
Corophium insidiosum	F	F	C	C	C	P	P	P	
Gammarus salinus	F	P	F	F	F	F	P	P	P
Jassa marmorata	P					P			P
Microdeutopus gryllotalp	F	F	C	C	C	P	P	P	P
Bryozoa		√	√	√	√		√	√	
Balanus improvisus				√		√	√	√	√
Carcinus maenas		√							
Nereis spp.	√	√	√	√	√		√	√	√
Unid scale worm								√	√
Molgula manhattensis				√	√			√	
Mytilus edulis	√	√	√	√	√	√	√	√	√
Mytilus juv. <15mm	√	√	√	√	√	√	√	√	√

	Albert Dock May1990				Albert Dock August 1990			
Depth in m	0.2	1	2	3	0.2	1	2	3
Corophium insidiosum	P	P	F	F	F	C	A	C
Gammarus salinus	F	F	F	F	P	P	P	F
Jassa marmorata	P				F			
Microdeutopus gryllotalp	P	F	F	F	C	C	C	C
Bryozoa	√	√	√		√			√
Obelia dichotoma	√	√						
Balanus improvisus	√		√	√				√
Nereis spp.		√	√	√		√		
Polydora spp.		√		√				
Botryllus schlosseri		√			√			√
Molgula manhattensis			√	√	√	√	√	√
Mytilus edulis	√	√	√	√				
Mytilus juv. <15mm		√	√	√	√	√	√	√
Littorina saxatilis	√							

	Albert Dock January 1991			
Depth in m	0.2	1	2	3
	Amphipods not enumerated.			
Corophium insidiosum	√	√	√	√
Gammarus salinus	√	√	√	√
Jassa marmorata	√			
Microdeutopus gryllotalp	√	√	√	√
Halichondra panicea			√	√
Balanus improvisus		√		√
Nereis spp.			√	
Ascidiella aspersa				√
Botryllus schlosseri		√		
Ciona intestinalis		√	√	√
Mytilus edulis	√	√	√	√
Mytilus juv. < 15 mm		√	√	√

Amphipod abundance estimates:
Abundant (A) = >1000
Common (C) = 100 - 999
Frequent (F) = 10 - 99
Present = < 10
Animals per 25 x 25 cm quadrat.
Other species presence /absence only.

Notes: Nereis - species present are N. virens, N. pelagica, N. diversicola.
Polydora - Mainly P. ciliata, occ. P.pulchra.

App Table XVII Queens Dock - Fauna in 25 x 25 cm Wall Scrape.

	Queens Dock June 1988			Queens Dock July 1989		
Depth in m	0.2	1	2	0.2	1	2
Corophium insidiosum	√	√	√	F	C	C
Gammarus salinus	√	√	√	P	P	F
Jassa marmorata				P	F	F
Microdeutopus gryllotalpa	√	√	√	F	C	C
Bryozoa	√	√	√		√	√
Balanus improvisus					√	√
Nereis spp.	√					√
Molgula manhattensis					√	√
Mytilus juv. <15mm			√		√	√

	Queens Dock Dec.1989			Queens Dock May 1990			
Depth in m	0.2	1	2	0.2	1	2	3
Corophium insidiosum	C	C	A	C	A	A	C
Gammarus salinus	P	P	P	P	P	P	P
Jassa marmorata	C	F	F	C	C	C	
Microdeutopus gryllotalpa	F	F	C	F	C	C	C
Conopeum seurati	√	√	√	√	√	√	
Obelia dichotoma				√	√	√	
Balanus improvisus	√	√		√	√	√	√
Nereis spp.		√	√	√	√	√	
Polydora spp.	√	√	√	√	√	√	√
Botryllus schlosseri				√	√	√	
Molgula manhattensis	√	√	√			√	
Mytilus edulis	√	√	√		√	√	√
Mytilus spat				√	√	√	√

	Queens Dock August 1990			
Depth in m	0.2	1	2	3
Corophium insidiosum	A	A	A	A
Gammarus salinus				
Jassa marmorata	C	C	C	C
Microdeutopus gryllotalpa	A	C	A	A
Bryozoa	√	√	√	√
Balanus improvisus		√	√	
Capitella captata		√	√	
Nereis spp.		√	√	√
Polydora spp.		√	√	√
Molgula manhattensis	√	√	√	√
Mytilus edulis		√	√	√
Mytilus juv. < 15mm	√	√	√	√

Key -
Amphipod abundance estimates:
Abundant (A) = >1000
Common (C) = 100 - 999
Frequent (F) = 10 - 99
Present = < 10
Animals per 25 x 25 cm quad.
All other species presence/ absence only.

Bold ticks indicate dominant amphipod species where animals not counted.

Key -
Nereis, species present are - N. virens, N. pelagica, N. diversicola.
Bryozoa -Mainly Conopeum seurati
Polydora - Mainly P. ciliata, occ. P.pulchra.

Table XVIII

	August 1989			August 1990		
Fauna observed in 5 minute search.	Graving	Albert	Queens	Graving	Albert	Queens
Bryozoa						
Halichondria panicea					√	
Metridium senile			√	√		
Sagartia troglodytes	√		√	√	√	√
Urticina felina		√				√
Carcinus maenas	√	√	√	√	√	√
Crangon crangon		√				
Praunus flexuosus		√		√	√	√
Palaemonetes varians	√					
Botryllus schlosseri		√			√	√
Ciona intestinalis				√	√	
Molgula manhattensis	√	√	√	√	√	√
Styela clava					√	
Mytilus edulis	√	√	√	√	√	√
Anguilla anguilla		√				
Chelon labrosus		√				
Gasterosteus aculeatus				√		
Platichthys flesus		√				
Pleuronectes platessa		√				
Pomatoschistus microps	√	√				√
Sprattus sprattus	√					